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Influence of climatic gradients on metacommunities of aquatic invertebrates on granite outcrops in southern Western Australia

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Some 210 pools on 21 rock outcrops arranged along two climatic gradients were surveyed for invertebrates in the winter–spring of 2010 and 2011. Almost all pools had clear fresh waters and most were near-full when sampled. Ninety-three taxa were identified, with crustaceans and insects about equally represented; crustaceans were numerically dominant, particularly *Limnadia badia*, *Boeckella opaqua* and *Cypericercus* spp. Individual rock metacommunities ranged from 19 to 35 species, with notable decreases along a southwest–northeast climatic gradient, but little decrease along an east–west transect where climatic changes were less. Species richness was also influenced by the relative number of pools on each rock, with two saline pools added to one rock's metacommunity causing an anomaly. Diversity of cladocerans decreased most along the gradients with ostracods next and insects hardy affected. The most important influencing factor is hydroperiod which is largely determined by an interaction between rainfall and evaporation. In stress years such as 2010, these are reduced and some species may not complete their life cycle. Future climate change may act to reduce hydroperiods, particularly at northern and northwest fringes of the Wheatbelt and Goldfields.

KEYWORDS: aquatic insects, branchiopods, cladocerans, climate, gnammas, hydroperiod, ostracods.

INTRODUCTION

In recent decades there have been a number of studies on rock pools (pan gnammas *sensu* Twidale & Corbin 1963) on granite outcrops in southern Western Australia, focused on the Wheatbelt and adjacent Goldfields regions (Bayly 1982, 1997; Pinder *et al.* 2000; Jocqué 2007; Timms 2006, 2012; Jocqué *et al.* 2007). Overall, they reported a rich aquatic fauna, with a large endemic crustacean component. Individual pools are small and rocks isolated so any pool or set of pools supports only part of the total metacommunity known to occur in gnammas (Jocqué *et al.* 2010). As well as each pool/pool set having a variable composition due to stochastic events and local environmental factors, there is evidence of regionalisation of the fauna (Timms 2012). The Northcliffe pools (near the coast in far southwestern Australia) studied by Bayly (1982) have but a small subset of species common across the Wheatbelt and moreover there is a decline in species richness in Wheatbelt transects south-southeast to north-northwest (Jocqué 2007; Timms 2012). It is possible both differences could be associated with climatic influences expressed mainly in hydroperiod lengths determined by the interaction of seasonal rainfall and evaporation. In addition, different energy pathways in the Northcliffe pools could be contributing factors (Bayly 1982; Timms 2012).

It is likely there are other differences between pool groups in different districts, with field experiences suggesting widespread lower species richness in gnammas at the western and southern fringes of the Wheatbelt and towards the Goldfields and non-agricultural land to the north and northeast, as well as metacommunity differences in rock pools near the coast. As in the case of the Northcliffe series, those on rocks at the eastern fringe at the Arid Woodland – Nullarbor interface, namely at Balladonia, seem to have some species not found elsewhere (Timms 2006). It is the aim of this study to examine pools on rocks on the northeast and eastern edges of the Wheatbelt and onwards to the Mulga association in the north and Arid Woodland in the east in order to detect differences in diversity and species composition from that pertaining in the area of perceived maximum diversity near Hyden. Although diversity may be a little higher to the south of here (e.g. at Mt Madden: Timms 2012), the Hyden area's gnammas are the best studied and so provide a suitable reference area. It is possible other factors such as pool size, numbers and salinity influence metacommunity diversity (Jocqué *et al.* 2010; Vanschoenwinkel *et al.* 2009; Waterkeyn *et al.* 2008), so note will be taken of these.

METHODS

Pan gnammas vary greatly in size, although generally the pools are shallow (<25 cm deep), flat floored, somewhat circular and <5 m in diameter (Twidale & Corbin 1963; Timms 2012). The rock outcrops containing them are highly variable both in size and elevation above the surrounding plain and also the number of pools per outcrop varies widely.

The present study is concerned with transects in different directions, one to the northeast and one to the east of Hyden: the first across a rainfall gradient of 342–250 mm mean annual rainfall and 350 km long, and the second parallel to isohyets once east of Hyden and in a similar rainfall regime of ~260 mm annual rainfall (264

mm at Norseman and 263 mm at Balladonia) (Figure 1) and extending beyond the Arid Woodlands of the southern Goldfields to the edge of the Nullarbor, a distance of 560 km.

In this study of rock outcrops along two transects, three rocks in each area were chosen instead of one (as in Jocqué 2007 and Timms 2012) in order to facilitate comparisons statistically. In establishing a transect, granite rock outcrops should ideally be equidistant apart, be equal size and shape, and have equal numbers of similarly sized pools. This was not always possible because of environmental heterogeneity, but nevertheless a measure of homogeneity was achieved. Rock areas averaged 133 km apart with greatest irregularity caused by the Norseman set (rocks 13–15) only 95 km away from the Johnson set (10–12) and 200 km from the Balladonia set (16–18). Rock outcrops varied from being compact and well elevated (e.g. King Rocks, McDermid Rocks) to being of broken exposure and virtually at surrounding ground level (e.g. Afghan Rocks, Old Rainey Rocks), but these differences were not thought significant. Of more importance is the number of pools per rock, as this could influence species richness in a rock pool set (B Vanschoenwinkel pers. comm. 2012); while most rock outcrops had 50–100 pools, some had only a few more than those sampled (e.g. Ularring Rock with 15) or many more (e.g. Hospital Rocks with > 200) (Table 1). Ten pools were sampled per rock, a number which gives an acceptable representation of aquatic diversity (Pinder *et al.* 2000; Jocqué *et al.* 2007; Timms 2012). Because species richness is influenced by pool size (Vanschoenwinkel *et al.* 2009) it was important to use a standard ratio of small (<1.9 m) diameter to medium (2–5 m) and large (>5.1 m). The ratio of 2:6:2, small/medium/large, was chosen because medium-sized pools were the most common on

almost all the rocks chosen. Generally, the average diameter for each pool set on each rock was between 4 and 5 m (Table 1), though median values were 1–1.5 m lower because averages were skewed by some very large pools (10–20 m diameter). Ularring Rock had unusually small pools and Hospital Rocks had the biggest pools (Table 1). Field work was in August and early September when pools are generally at or near maximum depths and have maximal biotic diversity (Timms 2012). The range of individual pool depths was 5–36 cm, the lower value by design (pools <5 cm depth were rejected) and most averages were between 6 and 14 cm. Most rocks were visited in 2011, with some data extracted from an earlier study in 2010 (Timms 2012). While almost all rocks were visited just once, some longer term comparative data are available for the Balladonia series (rocks 16–18) (B V Timms unpubl. data), for Hyden and King Rocks (Jocqué *et al.* 2007), and seasonal data for the five rocks on a north-northwest–south-southeast transect (Timms 2012).

Sampling protocol was similar to that used by Timms (2012) on five rocks on a north-northwest transect used as comparative data in this study. Pool water depth was determined with a stiff tape measure at the deepest point to the nearest centimetre and pool size was estimated with the same tape by measuring two cross sections at right angles to nearest 5 cm. Conductivity was measured with a Hanna HI8733 meter, and general observations made on turbidity using a Waterwatch turbidity tube.

Two different nets were used to sample the fauna. A small plankton net, aperture 15 x 10 cm, 30 cm long and of mesh 159 µm, was used to sample microfauna in the water column and on the bottom (the latter by disturbing the sediment and then passing the net through the

Table 1 Rocks studied, location, size and number of pools.

| No.	Rock name	Rock set name	Coordinates	Date visited	Average diameter of pools (m)	Average depth of pools (cm)	No. of pools
1	Hyden Rock	Hyden	32°26'44"S, 118°54'14"E	1 Aug 2010	4.16	10.8	~50
2	Anderson Rocks	Hyden	32°10'03"S, 118°51'11"E	26 Aug 2010	3.84	8.5	~100
3	King Rocks	Hyden	32°19'00"S, 119°09'09"E	27 Aug 2010	5.74	14.2	~100
4	Boondi Rock	Highway	31°10'45"S, 120°22'56"E	4 Aug 2011	5.95	12.6	~50
5	Boorabin Rock	Highway	31°12'13"S, 120°17'20"E	4 Aug 2011	4.65	13.4	~50
6	Karalee Rocks	Highway	31°09'47"S, 120°32'55"E	5 Aug 2011	4.82	10.6	~100
7	Old Rainey Rocks	Menzies	29°43'37"S, 119°37'25"E	5 Aug 2011	3.38	7.5	~25
8	Hospital Rocks	Menzies	29°50'20"S, 120°07'06"E	6 Aug 2011	8.73	12.7	~200
9	Ularring Rocks	Menzies	29°55'32"S, 120°32'56"E	6 Aug 2011	4.04	9.5	~15
10	McDermid Rock	Johnson	32°01'16"S, 120°44'13"E	30 Aug 2011	4.36	10.4	~75
11	Banks Rock	Johnson	31°55'48"S, 120°15'58"E	31 Aug 2011	4.75	10.9	~40
12	Disappointment Rock	Johnson	32°07'56"S, 120°55'38"E	31 Aug 2011	3.92	6.7	~15
13	McPherson Rocks	Norseman	32°27'10"S, 121°40'28"E	1 Sep 2011	4.22	6.0	~20
14	Unnamed rocks	Norseman	32°24'56"S, 121°40'51"E	2 Sep 2011	3.81	5.5	~40
15	Theatre Rocks	Norseman	32°08'25"S, 121°33'23"E	2 Sep 2011	4.73	5.3	~40
16	Afghan Rocks	Balladonia	32°21'23"S, 123°40'05"E	3 Aug 2011	5.67	8.0	~40
17	Booanya Rock	Balladonia	32°45'38"S, 123°36'27"E	2 Aug 2011	5.56	9.7	~30
18	Balladonia Rock	Balladonia	32°27'39"S, 123°51'52"E	2 Aug 2011	3.15	7.9	~20
19	Mt Madden	–	33°14'23"S, 119°50'34"E	1 Aug 2010	4.12	10.7	~150
20	Yanneymooning Rock	–	30°42'50"S, 118°33'19"E	2 Aug 2010	4.13	6.8	~120
21	Bullamanya Rocks	–	29°09'51"S, 117°39'40"E	3 Aug 2010	5.01	5.5	~150
22	Walga Rock	–	27°24'14"S, 117°27'49"E	8 Sep 2010	4.00	9.8	~20

slurry). A household sieve, 20 cm in diameter and of mesh size ~2 mm was used for larger invertebrates. The number (5–20) of sweeps of each device was adjusted to pool size so that about 50% of the water volume in small pools was sampled down to 5% in the very large pools. This introduced an error between pools in estimating abundance and species richness, but the bias was constant between rocks.

Plankton were concentrated by settling first in dilute ethanol, then pouring off the supernatant and preserving in >70% ethanol. Sieve collections were handpicked in the field from a sorting tray; one individual (a few of more abundant chironomids) of each species was kept and preserved in 70% ethanol and the remainder returned alive to the pond. In the laboratory each plankton collection for each pool was examined completely under an Olympus SZ61 stereomicroscope, species identified and the number of each species estimated on a log scale. The number of macroinvertebrates caught in the sieve was also estimated on a log scale. In both cases an index of abundance of each species in the 10 pools was calculated by adding abundances in each pool. Also in determining species richness for a rock, beetle larvae are not counted unless the adults were not recorded.

The term metacommunity is here used in two senses: to describe the whole set of animals found on an individual rock [the community in each pool contain only a subset of the animals living in all pools on a rock (Vanschoenwinkel *et al.* 2007; Jocqué *et al.* 2010)] or to describe the communities living on three rocks close to one other. The rocks studied were chosen to form six sets: Hyden, Highway and Menzies on the northeastern transect and Hyden, Johnson, Norseman and Balladonia on the eastern transect (Figure 1; Table 1).

Statistical differences between rock areas were tested with two tailed t-tests. Relationships between assemblages on the five rocks were investigated using PRIMER (v5) (Clarke & Gorley 2001). For the north-northwest transect the average seasonal abundance data of invertebrates from the mid-July, early and late August and mid-September trips (Table 1) was log (N+1) transformed prior to multivariate analysis. In the northeastern and eastern transects the data collected on the single visits was similarly transformed prior to analysis. Non-metric multidimensional scaling, based on the Bray–Curtis similarity index was used to represent assemblage composition in two-dimensional space. Relative distance apart in the ordination represents relative dissimilarity.

Figure 1 Map of study area in inland Western Australia. Rock outcrops 1 to 22 as in Table 1. Three transects shown: north-northwestern from 19 to 22 (from Timms 2012), northeastern from 1,2,3 to 7,8,9, and eastern from 1,2,3 to 16,17,18. Major towns and roads and smoothed isohyets shown.
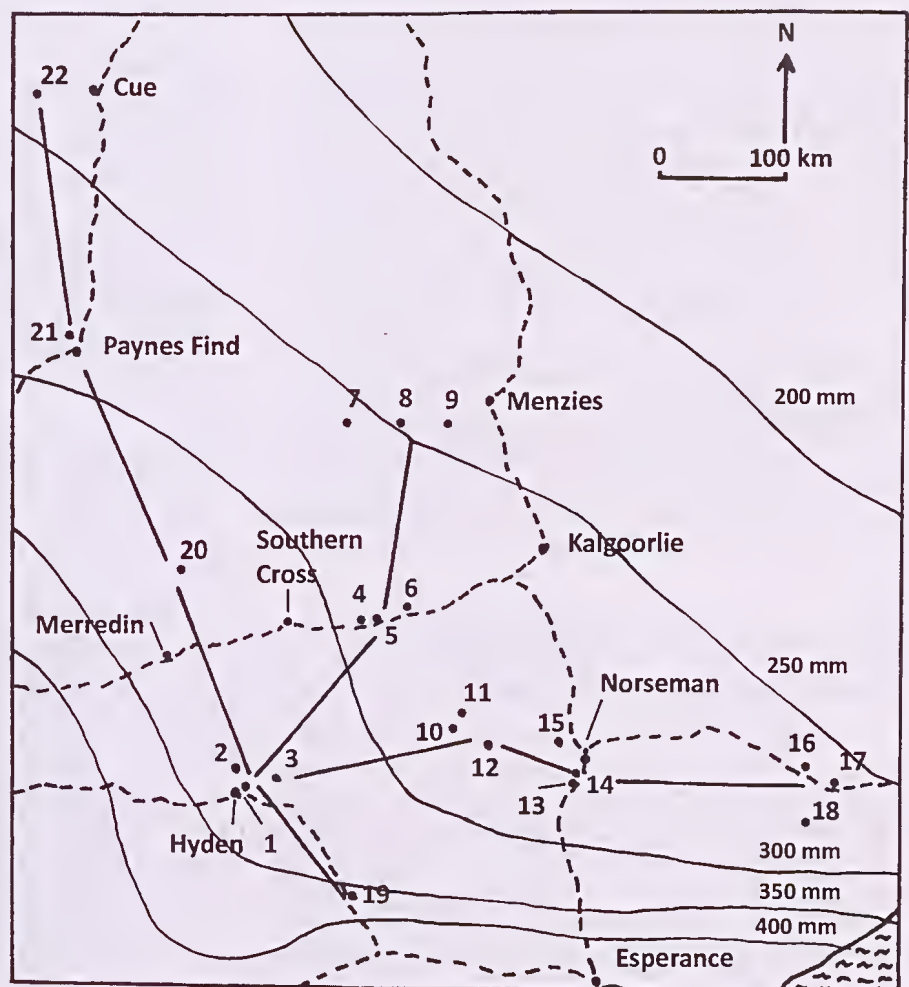
RESULTS

Most pools were full or near full when sampled as there had been adequate rainfall in the previous weeks. Conductivities were <200 µS/cm and often <75 µS/cm in all pools except two. Both exceptions were deeply incised pools on Booanya Rock, one with a conductivity of 4200 µS/cm and the other a conductivity of 9000 µS/cm; in both pools staining of the rocks suggested no overflow ever. All pools had turbidities less than 15 NTU, often lower, so that their floors were always visible.

The eastern transect yielded 78 taxa of invertebrates and the northeastern transect had 65 taxa (Appendices 1, 2), making 93 together. If full identifications were possible, the lists would be about 5% longer as a few taxa are thought to be comprised of two or more species. Dominant taxa on most rocks included *Limnadia badia*, *Boeckella opaqua* and *Cypericercus* spp. Other common species across many rocks or particularly common on just a few rocks were two unidentified planarians, *Branchinella longirostris*, *Caenestheriella mariae*, *Ceriodaphnia* sp., *Ilyodromus amplicolis*, *Stenopriscus multimaculatus* and *Paraborniella tonnoiri*. While crustaceans and insect species were similarly diverse on most rocks, it was the crustaceans which were by far the most numerous (Appendices 1, 2)

Momentary species richness (MSR) on individual rocks ranged from 19 to 35 species, with averages per rock set varying from 22.3 to 32.3 (Figure 2; Table 2). There was a small decrease in species richness from west to east along the eastern transect and a much larger decrease along the northeastern transect (Figure 2). Decrease eastwards along transects was due mainly to cladocerans and to a lesser extent to ostracods; insects remained just as diverse northeastwards and actually increased mid-length along the eastern transect (Figure 3). These changes were only significant between the Hyden and Highway rock sets, Hyden and Menzies sets and between the Johnson and Menzies rock sets (Table 3), thus confirming the distinctiveness of the two outlying northeastern sets.

Three rock sets had higher coefficient of variations than the others, due to one or more of the rocks having markedly higher or lower MSRs (Table 2). In the Menzies set, Hospital Rocks were most speciose coincident with this rock having many more pools than the other two; on the other hand in the Johnson set, Disappointment Rock was less speciose than its two nearby rocks coincident with this rock having comparatively fewer pools. In the

case of the Balladonia set, Booanya Rock was most speciose, possibly associated with it having two saline pools and hence some species unique to it. Also in this the most variable of all sets, Balladonia Rock had the lowest species richness and coincidently it had the fewest pools.

Distinctiveness of rock sets was even greater when explored using Bray–Curtis dissimilarities to create a multidimensional ordination (Figure 4). Rocks within sets were clumped closer together than between sets, indicating a strong geographical influence on pool fauna. Among the rock sets there were large distances between the three sets on the northeastern transect, but on the eastern transect the Johnson and Norseman sets clumped nearer each other and to the Hyden set while the Balladonia set was the most divergent of all. While some of this inequality is explained by the geographical closeness of the Johnson and Norseman sets and remoteness of the Balladonia set (see Figure 1), the Balladonia set contained a few species unique to it, including *Branchinella spinosa*, *Triops* n. sp., *Daphniopsis queenslandicus*, *Cabanocypris nunkeri* and *Heterocypris* sp. By contrast the other outlying sets had largely somewhat depauperate associations of species compared to the richer Hyden set (Appendices 1, 2).

DISCUSSION

Species richness figures for pools on rock outcrops depend on the collection methods and especially the number of pools sampled, either momentary or cumulative with time. For instance at Wave Rock the cumulative species count was still increasing even after 57 pools were sampled (Jocqué *et al.* 2007). Sampling such a large number of pools takes days and so is impractical, so generally studies sample just 10 pools, but this catches only about 80% of the species comprising the metacommunity and samples from just a few pools per rock catch an even lower percentage of species, e.g. five pools yielded ~50% of the species at Wave Rock (Jocqué *et al.* 2007). Regular seasonal studies add more species: e.g. at Wave Rock the present study reports 30 species, yet Timms (2012) found 41 species in a seasonal study,

Table 2 Species richness on the 6 rock sets.

Rock set	Value 1	Value 2	Value 3	Mean	SD	SE	C of V
Hyden	30	33	34	32.3	2.08	1.2	0.064
Johnson	35	33	28	32	3.61	2.08	0.113
Norseman	26	30	30	28.7	2.31	1.33	0.080
Balladonia	28	24	33	28.3	4.51	2.6	0.159
Highway	26	28	26	26.8	1.15	0.66	0.049
Menzies	19	27	21	22.3	4.16	2.4	0.186

Note that with a rock set, the richness values represent rocks in the order they are given in Table 1.

Table 3 t-tests on differences in MSR between grouped sites (significant values in bold).

Rock sets	P value
Hyden & Johnson	0.898
Hyden & Norseman	0.111
Hyden & Balladonia	0.257
Johnson & Norseman	0.270
Johnson & Balladonia	0.333
Norseman & Balladonia	0.916
Hyden & Highway	**0.026**
Hyden & Menzies	**0.034**
Highway & Menzies	0.224
Johnson & Highway	0.135
Johnson & Menzies	**0.038**
Norseman & Highway	0.272
Norseman & Menzies	0.104
Balladonia & Highway	0.598
Balladonia & Menzies	0.165

Figure 2 Number of species in metacommunities along the northeastern and eastern transects.
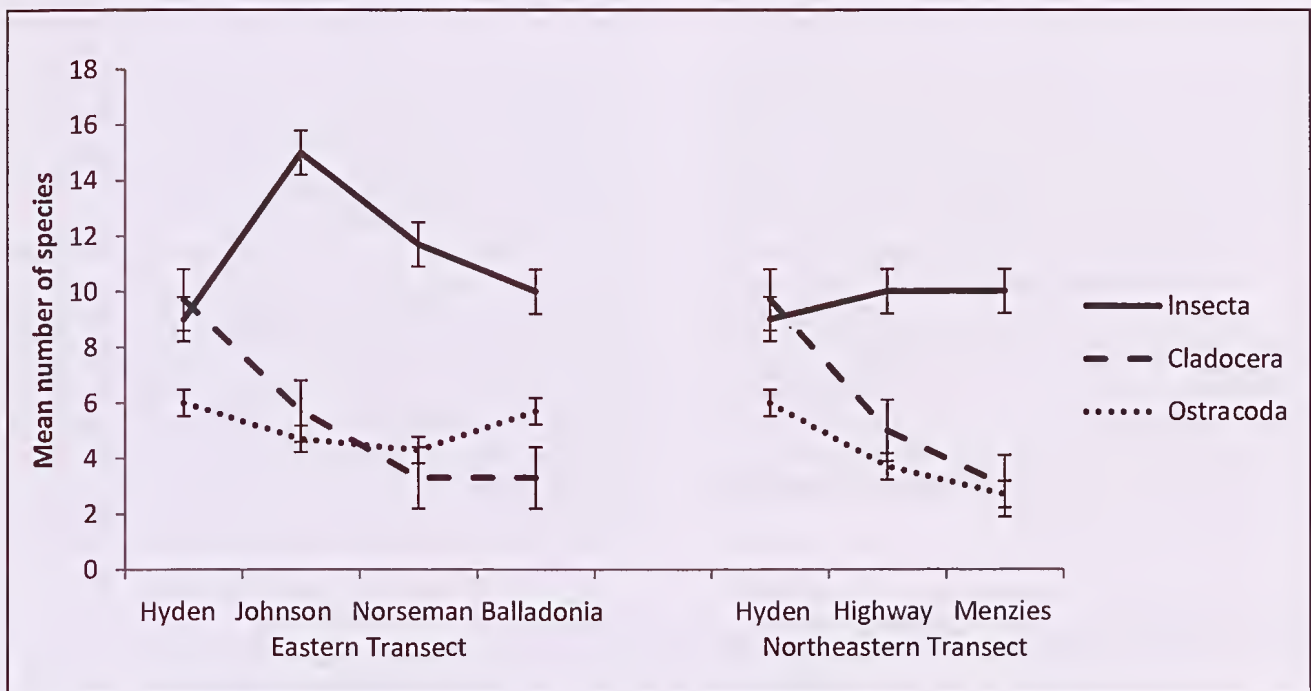
Figure 3 Changes in the number of insects, cladocerans and ostracods along the northeastern and eastern transects.

using exactly the same collecting methods. Studies over many years can further extend lists, with the list for Balladonia increasing from 24 in this study to 33 over three years and with some species varying greatly in numbers from year to year (B V Timms unpubl. data). Hence in comparing data it is important to know the number of pools sampled and how often.

The extensive and detailed study by Pinder *et al.* (2000) of about 10 pools on each of nine rock outcrops and the survey of 36 pools by Bayly (1997) across the Wheatbelt revealed a weak areal pattern in metacommunities across the Wheatbelt, though the large branchiopod component showed no significant distributional pattern across the Wheatbelt and adjacent

Figure 4 Ordination diagram of the relationships between rock sets along the northeastern transects (thin line) and eastern transects (thick line). Individual rock numbers as listed in Table 1.
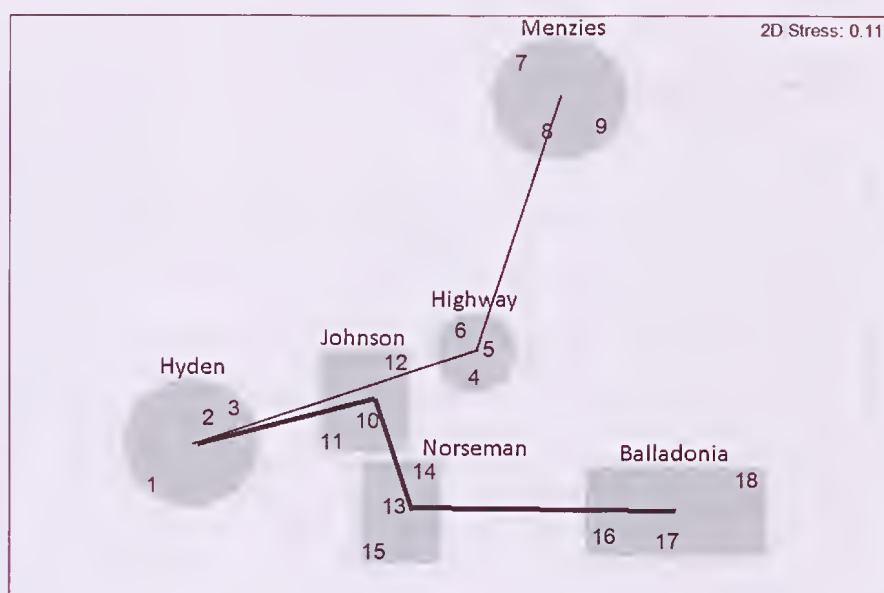
Goldfields (Timms 2006). However an examination of Bayly (1982), who studied rock pools near Northcliffe, near Albany and southeast of Perth (i.e. pools in coastal forests), revealed a restricted fauna compared to that in the Wheatbelt. Six Northcliffe pools had a metacommunity of just 13 species and lacked large branchiopods, insects and planarians and had only two ostracod and five cladoceran species. Seven pools southeast of Perth yielded 16 species, almost all common throughout the Wheatbelt, but nevertheless a depauperate metacommunity compared to those on Wave Rock and King Rocks (Jocqué *et al.* 2007). Following on from an indication of differences between metacommunities in the northern, central and southern Wheatbelt (Pinder *et al.* 2000), in a north-northwest–south-southeast transect 700 km long from Wave Rock (near Hyden) to Walga Rock (west of Cue) (Figure 1), Jocqué (2007) noted a significant decrease in generic richness northwards with the metacommunities differing significantly from each other, with 26.7% of the variance explained by precipitation differences between north and south. More detailed studies along a similar transect (but with a slightly different array of rock outcrops), showed a similar decrease in species richness northwards (Table 4) (Jocqué 2007; Timms 2012).

In the present study, the decrease in species richness along the northeast transect is less than on the longer north-northwest transect of Jocqué (2007) and Timms (2012), and minimal on the eastern transect (Figure 2; Table 4). In that all three transects terminate in areas with ~250 mm rainfall, environmental factors other than rainfall annual averages may be influential. Two important ones could be evaporation and rainfall reliability. Annual evaporation is much greater at the northern terminus (~3000 mm at Cue) than at the northeastern Menzies terminus (2800 mm) and least at the eastern Balladonia terminus (2000 mm) (all figures estimated from Bureau of Meteorology 2012a). Such differences mean that pool water persists much longer further south and hence fauna have a better chance of completing life cycles at southern sites. Rainfall is almost twice as reliable in the south at Balladonia than at Cue, with Menzies intermediate (Gaffney 1975). Again, but over a longer term, rainfall unreliability might adversely influence metacommunities in the north by disrupting life cycles and by reducing survivorship of propagules (Boulton & Lloyd 1992; Jenkins & Boulton 2007). For instance the drought year of 2010 was far more disruptive of life cycles at Walga Rock in the north that at rocks in the south (Timms 2012). These are aspects of

Table 4 Data on variation in metacommunities along a NNW–SSE transect.

	Mt Madden	Hyden Rock	Jilbadgie Rock	Beladgie Rock	Elach-butting	Remlap Rock	Bullamanya Rocks	Walga Rock
Jocqué (2007)								
Generic richness	–	33	30	24	22	37	20	13
Timms (2012)					Yanney-mooning			
Insecta	17	17	–	–	14	–	16	10
Cladocera	15	13	–	–	11	–	12	0
Ostracoda	8	7	–	–	6	–	7	2
Total spp.	51	41	–	–	41	–	37	16

the overall hydroregime, a factor of paramount importance in influencing community structure of rock pools (Vanschoenwinkel *et al.* 2009).

A multiple regression of the above three factors gave the equation:

$$SR = 28.48 + 53.68x - 74.23y - 5.79z$$

where SR is species richness, x is rainfall in metres, y is evaporation in metres and z is a reliability factor based on coefficient of variation (Cowling *et al.* 2005). The equation explained 85% of the variance. From this, all three factors are influential (and rainfall and evaporation are significant at 05% level individually, with rainfall reliability significant at the 10% level). Evaporation needs to be added to rainfall as a factor influencing hydroperiod and hence species richness in pools. Rainfall reliability is most likely acting through differential survivorship of propagules, but no work has been done on species inhabiting gnammas.

Other factors influencing species richness in the present rock sets include the unique additional presence of saline species at Booanya Rock which elevate species richness in the Balladonia rock set (no other saline gnammas are known in southwest Western Australia: Bayly 1982, 1997; Jocqué *et al.* 2007; Pinder *et al.* 2000; Timms 2012). Probably also of influence is the total number of rock pools on each rock outcrop, increased numbers allowing for more colonisation and survival of species on a rock (B Vanschoenwinkel pers. comm. 2011). While every effort was made to standardise this factor's influence in this study, it seemed to affect species numbers positively on Hospital Rocks compared to other Menzies rock set, and negatively on Disappointment and Balladonia Rocks compared to other rocks in their respective sets. Interestingly Pinder *et al.* (2000) had lower species richness than average on rocks (Styles near Grass Patch, Punaputin near Wagin) which I know have few pools compared to other rocks they studied.

Of the various taxonomic groups represented in gnammas, the most important are large branchiopods, cladocerans, ostracods and various insects. Insects are least affected by environmental factors associated with climatic factors, diversity being only slightly less in pools on rocks in the harsher north and northeast (Figure 3; Table 3). The reason for this may lie in insects being active dispersers and also being largely unaffected by hydrology of the pools. Exceptions include some chironomids, but *Paraborniella tonnoiri* and *Allotrissoides* spp have special life cycle adaptations to survive in temporary pools (Edward 1968; Jones 1971). The absence of insects in the Northcliffe pools and their scarcity in the forest gnammas east of Perth is less easy to explain, but there may be many other suitable habitats nearby which harbour them whereas suitable habitats in the remote inland are scarce, gnammas being the main reliable water and hence particularly attractive to dispersing insects. There are too few large branchiopod species to plot in Figure 3, but they are widely distributed in rock pools across the Wheatbelt and Goldfields (Timms 2006), though less conspicuous in the gnammas in forests near Perth and absent from the Northcliffe pools. Either the markedly acidic nature of these pools is inimical to large branchiopods or their food or perhaps eggs do not

readily disperse eastwards against the prevailing southwesterly winds.

It is the cladocerans and ostracods that are most affected by the imposition of harsher climatic factors of the north, northeast and far east of the Wheatbelt–Goldfields of southern Western Australia (Figure 3; Table 4). The increase in richness of ostracods on the eastern transect is an anomaly and is due to the addition of saline species at Booanya rock. Most data suggest that the majority of species can withstand a range of hydrological conditions so that diversity decreases only slowly northwards and it is only in the remote goldfields or far east on the Nullarbor fringe that many species are absent. Common species living in a wide variety of hydrological conditions include *Limnadia badia*, *Boeckella opaqua*, *Macrothrix hardingi* and most of the insects. Notable absences of cladocerans from remote areas include *Ceriodaphnia* n. sp., *Daphnia jollyi*, *Leberis aenigmatosa*, *Planicirclus alticarinatus* and *Plurispina* sp. No species of ostracods can be identified in this decrease, partly because they are less well known taxonomically but overall they may be more resilient than cladocerans.

Climate is changing and already over the last 41 years rainfall has decreased 2–10% in much of the wheatbelt (Bureau of Meteorology 2012b), though it is not known if this has affected invertebrate communities in rock pools by shortening hydroperiods. For most of the Wheatbelt it probably has not, as all crustaceans and almost all insects completed their life cycles in the drought year of 2010 at all rocks except at the most northern site, Walga Rock (Timms 2012). However a predicted further decrease of 2–5% in annual rainfall in southern Western Australia (Bureau of Meteorology 2012b) may at least curtail hydroperiods in the drier areas, disrupt life cycles and hence adversely affect diversity. The evidence for the most northern site, Walga Rock suggests the species list was almost exactly the same in 2009 (M Jocqué unpubl. data) as it was in the drought year of 2010 (Timms 2012), the difference between the two years being in the failure of many species to complete their life cycles in 2010. How often this can occur before local extinction takes place is not known and it is possible that some species still common further south may have already succumbed in the far north.

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APPENDIX 1 SPECIES LISTS FOR ROCKS ON NORTHEASTERN TRANSECT

Major group	Species	Hyden	King	Anderson	Boondi	Boorabin	Karalee	Old Rainey	Hospital	Ularring
Turbellaria	unidentified black planarian	xx	xx	xx	xx	xx	xx	xx	xx	xx
	unidentified green planarian	xx	x	x	-	-	-	-	-	-
	Unidentified rhabdocoel	-	-	o	-	-	-	-	-	-
Nematoda	unidentified nematodes	xx	xx	xx	x	o	o	xx	x	o
Branchiopoda	*Branchinella longirostris*	r	o	x	x	-	x	-	xxx	x
	Limnadia badia	xx	xx	xx	xxx	xx	xx	xx	xxx	xx
	Caenestheriella mariae	-	o	-	x	x	x	-	-	-
	Daphnia jollyi	-	x	-	x	x	x	-	-	-
	Ceriodaphnia sp.	xxx	xxx	xxx	-	-	o	-	-	-
	Moina australiensis	-	x	x	-	-	-	-	-	-
	Macrothrix hardingi	-	x	x	o	o	x	x	r	x
	Macrothrix spp.	-	x	x	-	-	-	-	x	-
	Neothrix spp.	x	x	x	r	o	o	x	xx	o
	Alona macrocopa	r	r	o	-	-	-	-	-	-
	Alona rigidicaudis	-	-	-	-	-	-	-	o	o
	Chydorus hybridus	x	-	x	-	-	-	-	-	-
	Ephemeroporus tridentata	x	x	x	o	o	o	-	-	-
	Leberis aenigmatosa	x	x	x	x	x	x	-	-	-
	Planicirculus alticarinatus	-	-	x	-	-	-	-	-	-
	Plurispina multituberculata	xxx	xx	xx	-	-	-	-	-	-
	Rak sp.	xx	-	-	-	-	-	-	-	-
Copepoda	*Boeckella opaqua*	xxxx	xxxx	xxxx	xx	xx	x	-	xxxx	r
Ostracoda	*Bennelongia* spp.	xx	x	xx	x	-	xx	-	-	-
	Candonocypris spp.	xx	x	x	x	x	x	-	-	x
	Cypretta baylyi	x	x	x	-	-	-	-	-	-
	Cypericercus spp.	xxx	-	xxx	xx	x	xxx	-	x	x
	Ilyodromus amplicolis	xx	xx	xx	xx	xx	xx	xx	xx	x
	Limnocytherae spp.	-	x	o	-	-	-	-	-	-
	Sarscypridopsis spp.	xx	-	xx	-	-	-	-	-	-
Odonata	*Hemicordulia tau*	-	-	-	-	-	o	r	o	-
Hemiptera	*Micronecta gracilis*	-	r	r	r	-	r	-	o	-
	Agraptocorixa parvipunctata	-	o	-	r	o	r	-	o	-
	Anisops gratus	-	r	-	-	r	-	-	r	-
	Anisops hyperion	-	-	-	-	r	-	-	-	-
	Anisops stali	-	-	-	-	-	r	r	-	-
	Anisops thienuemanni	r	-	o	r	o	r	-	x	-
Coeloptera	*Allodessus bistrigatus*	o	o	-	-	o	-	-	-	-
	Antiporus gilberti	-	r	-	-	r	-	-	-	-
	Berosus spp.	r	-	-	-	-	-	-	-	o
	Enochrus sp.	-	-	-	-	-	-	-	-	r
	Eretes australis	-	-	-	r	-	-	r	o	o
	Megaporus howitti	-	-	o	-	-	-	-	-	-
	Paroster sp.	-	-	-	x	o	xx	x	x	xx
	Rhantus suturalis	-	-	-	-	-	-	-	r	-
	Sternopriscus multimaculatus	r	x	x	xx	xxx	xx	x	xx	xx
	Allodessus larva	r	-	-	-	-	o	-	-	-
	Antiporus larva	-	-	-	-	-	-	o	-	-
	Paroster larva	o	o	-	x	o	x	x	x	x
	Sternopriscus larva	o	o	o	-	-	-	-	-	-
Tricoptera	*Triplectides australis*	-	-	-	-	x	-	o	o	-
	unidentified species	-	-	-	-	r	-	-	-	-
Diptera	juvenile chironomids	o	x	x	x	x	-	-	-	-
	Allotrissocladius sp.	x	o	o	-	x	-	r	xxx	xx
	Chironomus tepperi	-	-	-	-	-	-	-	o	-
	Paraborniella tonnoiri	x	x	x	xx	x	x	xxx	xxx	xx
	Parakeifferiella variagatus	xx	-	-	-	-	-	-	-	-
	Procladius sp. (Pentaneurini)	o	-	o	r	-	-	-	r	x
	Dasyhelea sp.	x	x	x	r	x	o	x	x	-
	Aedesalbo annulatus	-	o	-	-	x	-	x	-	o
	tipulid larvae	-	-	-	r	-	-	-	-	-
Mollusca	*Glyptophysa* sp.	-	o	,o	-	-	-	-	-	-
	Isidorella sp.	-	o	-	-	-	-	-	-	-
Arachnida	oribatid mite	-	-	-	-	-	-	o	-	-
	trombidioid mite	-	-	-	o	x	o	o	r	r
Momentary Species Richness		30	33	34	26	28	26	19	27	21

Species frequencies: r, 0.1; o, 0.2–0.9; x, 1–3: xx, 3.1–9.9; xxx, 10–19.9; xxxx, >20.

APPENDIX 2 SPECIES LISTS FOR ROCKS ON EASTERN TRANSECT.

Major group	Species	McDermid	Banks	Disappointment	McPherson	Unnamed	Theatre	Afghan	Balladonia	Booanya
Turbellaria	unidentified black planarian	x	xx	xx	xx	x	-	x	x	xx
	unidentified green planarian	-	xx	-	xx	xxx	x	x	-	o
Nematoda	unidentified nematodes	-	r	o	-	o	x	-	xx	o
Rotifera	Hexarthra sp.	-	-	-	-	-	-	-	-	o
Branchiopoda	Branchinella basispina	-	-	-	-	-	-	-	x	-
	Branchinella longirostris	x	xxx	x	o	xx	-	-	-	xx
	Limnadia badia	xxx	xx	xx	xx	xxx	xx	xx	xxx	xx
	Lynceus macleayanus	-	-	-	-	-	-	-	-	x
	Caenestheriella mariae	x	xxx	x	xx	xx	xxx	x	-	-
	Triops n. sp.	-	-	-	-	-	-	o	x	-
	Daphnia jollyi	-	-	-	-	o	-	-	-	-
	Daphnia queenslandicus	-	-	-	-	-	-	-	-	x
	Ceriodaphnia sp.	xx	xx	x	xx	xx	xxx	-	-	-
	Moina australiensis	-	-	-	-	-	x	x	-	x
	Macrothrix hardingi	o	-	xxx	r	-	-	-	-	-
	Macrothrix spp.	-	-	-	-	-	-	x	-	o
	Neothrix spp.	x	x	x	o	-	r	x	-	o
	Alona rigidicaudis	o	o	o	-	o	x	o	-	-
	Alona spp.	-	-	-	-	xx	-	-	-	-
	Celsinotum sp.	-	-	-	-	-	-	-	-	x
	Chydorus hybridus	o	o	-	-	-	-	-	-	r
	Ephemeroporus tridentata	xx	x	-	-	x	-	-	-	-
	Leberis aenigmatosa	xx	-	r	-	xx	-	-	-	-
	Plurispina multituberculata	xx	-	-	r	xx	-	-	-	-
Copepoda	Boeckella opaqua	xx	xxxx	xxx	xxx	xx	xxx	xxxx	x	x
	cyclopoid	r	-	-	-	-	o	o	r	-
	Harpacticoid sp 674 (DEC)	x	-	-	-	-	o	-	-	-
Ostracoda	Bennelongia spp.	-	o	x	x	x	x	-	-	xx
	Cabanocypris nunkeri	xx	-	-	-	-	-	-	xx	-
	Candonocypris spp.	-	xx	o	x	x	x	o	-	-
	Cypretta baylyi	xxx	-	-	-	-	-	o	-	-
	Cypericercus spp.	-	xxx	xxx	xx	xxx	xxx	xx	xx	xxx
	Cyprinotus edwardsi	xx	-	-	-	-	-	-	-	x
	Heterocypris sp.	-	-	-	-	-	-	-	-	xx
	Ilyodromus amplicolis	xx	xx	xx	x	xx	xx	xx	xx	xx
	Limnocytherae spp.	o	o	-	-	x	-	x	r	x
	Sarscypridopsis spp.	-	-	-	-	-	-	x	x	-

	Species								
Odonata	Hemicordulia tau	o	-	-	-	-	-	r	-
	Ischnura heterosticta	-	-	-	-	-	r	r	-
	Orthetrum caledonicum	o	-	-	-	-	-	-	-
Ephemeroptera	Cloeon sp.	-	o	-	-	-	-	-	-
Hemiptera	Micronecta gracilis	x	r	o	o	o	o	r	xx
	Agraptocorixa parvipunctata	o	r	r	o	o	o	-	x
	Anisops gratus	-	r	-	r	r	-	-	o
	Anisops hyperion	r	-	o	-	-	o	r	-
	Anisops stali	r	r	o	r	-	-	-	-
	Anisops thienemanni	o	-	-	-	o	x	r	xx
Coleoptera	Allodessus bistrigatus	x	-	r	o	o	-	-	-
	Antiporus gilberti	r	o	o	o	o	-	r	-
	Berosus spp. (mainly B. nutans)	o	o	o	r	r	o	r	-
	Cybister tripunctatus	r	r	-	-	-	-	-	-
	Enochrus sp.	-	-	-	o	o	-	-	-
	Eretes australis	o	o	r	o	o	o	o	o
	Hyphydrus sp.	-	o	o	o	o	-	-	-
	Limnoxenus zeakandicus	r	-	r	r	-	-	-	-
	Paroster sp.	o	x	-	-	-	-	-	-
	Sternopriscus multimaculatus	xx	xx	xxx	xxx	xx	xx	xx	xx
	Allodessus larva	-	-	-	-	-	-	-	-
	Eretes larva	-	-	-	-	-	-	-	o
	Paroster larva	x	x	x	x	x	x	x	-
Tricoptera	Triplectides australis	o	r	o	r	o	-	-	-
Diptera	juvenile chironomids	o	-	o	o	o	-	-	-
	Allotrissocladius sp.	o	-	x	-	-	-	-	-
	Chironomus tepperi	x	-	-	-	-	o	-	xx
	Paraborniella tonnoiri	x	xx	xx	xx	xx	xx	o	r
	Procladius sp. (Pentaneurini)	x	-	r	-	r	r	-	o
Mollusca	Dasyhelea sp.	o	x	x	x	o	o	xxx	o
	Glyptophysa sp.	-	-	-	-	-	-	-	o
	Isidorella sp	-	-	x	x	xx	-	x	-
Arachnida	hydrocarinid mite	-	-	-	-	-	-	-	r
	trombidioid mite	-	-	-	-	-	-	r	-
	Momentary Species Richness	35	33	28	26	30	30	28	2433

Species frequencies: r, 0.1; o, 0.2-0.9; x, 1-3; xx, 3.1-9.9; xxx, 10-19.9; xxxx, >20.

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Foraminifera from microtidal rivers with large seasonal salinity variation, southwest Western Australia *

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Large seasonal variations in salinity due to microtidal conditions and highly seasonal rainfall occur in the tidal reaches of the Serpentine River (51 maximum salinity variation), Murray River (38) and Collie River (32) which flow into large estuarine lagoons in southwest Western Australia. Preliminary observations are made on the distributions of foraminifera in these rivers that may reflect salinity changes. Thirty-five foraminiferal species are recorded from the rivers, including 10 species from the Serpentine River, 14 species from the Murray River and 33 species from the Collie River. Live foraminifera, determined by Rose-Bengal staining are very rare. The total (dead + live) foraminiferal assemblage is dominated by few species groups including hyaline *Ammonia* gr. *tepida*, *Elphidium* gr. *gunteri*, *E.* gr. *excavatum* and *E.* gr. *advenum*, porcelaneous *Quinqueloculina* gr. *seminula*, and organic-cemented agglutinated *Ammobaculites exiguus*. Among the live hyaline foraminifera, *Ammonia* gr *tepida* shows the greatest tolerance to salinity variation, occupying the entire 51 range recorded in the rivers. Live *Elphidium* gr. *gunteri*, *E.* gr *excavatum* and *E.* gr. *advenum* are found only at salinities less than about 44, extending to salinities of below 1. The dominant porcelaneous group, thin-walled small *Quinqueloculina* gr. *seminula*, lives in the rivers through a salinity range of 24–45 and shows progressive decrease in test size upstream from the river mouths. The agglutinated *Ammobaculites exiguus* is only common in the Serpentine River where it probably tolerates a very large salinity range. In the Murray and Collie Rivers organic-cemented agglutinated foraminifera are rare; and carbonate-cemented agglutinated forms are absent from all the rivers. Apart from the dominant species, most of the other hyaline and porcelaneous species seem to be confined, in both the dead and live assemblages, to the lower reaches of the rivers, where most live representatives are present under salt-wedge conditions approaching normal-marine salinities.

KEYWORDS: Collie River, foraminifera, Murray River, salinity distribution, Serpentine River, Western Australia.

INTRODUCTION

Lower reaches of rivers located in areas with highly seasonal rainfall and microtidal (<2 m) conditions are influenced by extreme seasonal variations in water quality due to variable freshwater outflow and the tidal forcing of inflowing marine water. This variability affects salinity, temperature, dissolved oxygen, light penetration, turbidity, suspended matter, nutrients and pH within the tidal reaches of the rivers and changes throughout the year. Because of warm dry summers and cool wet winters, coastal rivers of southwest Australia, particularly those that flow into large estuarine lagoons, display these characteristics (Brearley 2005; de Lestang *et al.* 2003; Gerritse *et al.* 1998; Kurup *et al.* 1998; McComb *et al.* 1998; Stephens & Imberger 1996).

Foraminiferal distributions in marginal-marine environments vary according to salinity, although other factors may also influence the distribution patterns (Murray 1991, 2006). Worldwide, there have been few studies of foraminiferal distributions in tidal rivers with seasonal salinity variations greater than 30 (Debenay *et al.* 1989, 2006; Murray 1991, 2006; Reddy & Rao 1984). In southwest Australia, the microtidal lower reaches of the Serpentine River, which flows into a large estuarine lagoon (Peel Inlet: Figure 1), recorded winter wet-season to summer dry-season variation in salinity of up to 51 during 2005–2006. Also within the region, during the same period, the microtidal reaches of the Murray River (flowing into Peel Inlet: Figure 1) and the Collie River (flowing into Leschenault Estuary: Figure 1) experienced large but less extreme salinity variations (up to 38 and 32 respectively). These rivers are excellent sites to study the influence of seasonal salinity changes on foraminiferal distributions in microtidal rivers whose entrances are located at varying distances across shallow estuarine lagoons from the open-marine shelf.

The aim of this paper is to make preliminary observations on the foraminiferal distributions in the microtidal Serpentine, Murray and Collie Rivers that flow into large estuarine basins in southwest Western Australia. Very large seasonal changes in salinity characterise the rivers and doubtless have a major influence on the fauna. This study adds significant data to knowledge of estuarine foraminifera in this part of Australia and will eventually contribute, when faunas from other microtidal rivers in the region become known, to more detailed biogeographic comparisons with estuarine foraminifera elsewhere.

REGIONAL SETTING

Geomorphology, freshwater inflow and salinity

The study area is located on the Swan Coastal Plain in southwest Western Australia (Figure 1). The Peel–Harvey Estuary is located on the western edge of the Swan Coastal Plain, about 80 km south of Perth (Brearley 2005), and consists of two interconnected coastal plain lagoons, the Peel Inlet and the Harvey Estuary (Gabrielson & Lukatelich 1985) (Figure 1). The total area of the Peel–Harvey Estuary is just over 135 km², and the Peel Inlet and Harvey Estuary are the largest inland water bodies in southwest Western Australia (Brearley 2005). The three rivers that flow into the Peel–Harvey Estuary are the Murray and Serpentine Rivers, located to the

Figure 1 Map of the Swan Coastal Plain in southwest Australia showing the location of studied rivers.

northwest of the Peel Inlet, and the Harvey River, located to the south of Harvey Estuary. The total catchment area of Peel–Harvey Estuary is 11 300 km², and total freshwater inflow to the estuary averages 430 x 10⁶ m³/year (Brearley 2005). The major source of freshwater is from the Murray River, with a catchment area of 6890 km² accounting for 60% of discharge. Around 12% of discharge into the Peel–Harvey Estuary occurs from the Serpentine River and 20% from the Harvey River (Brearley 2005).

The Leschenault Inlet (Figure 1) is an elongate shore-parallel lagoon located on the southern edge of the Swan Coastal Plain (Semeniuk et al. 2000). The estuary is 14 km long and 1.5–2.5 km wide, with a total area of about 27 km². The Collie River and the Preston River are the two river systems fluvially influencing the Leschenault Estuary (Semeniuk et al. 2000). Both rivers are located to the south of the estuary. The catchment area of the Leschenault Estuary is 4600 km². The Collie and Preston Rivers drain a total of 3600 km², although Wellington Dam traps the majority of the water. River discharge into the Leschenault Estuary occurs from a 770 km² area, of which 603 km² is drained by the Preston River (Brearley 2005). River discharge from the Collie River occurs from an area of only 167 km².

Artificial channels connecting both the Peel–Harvey and Leschenault Estuaries to the Indian Ocean have been constructed. The Dawesville Cut, a 2.5 km long channel located to the north of the Harvey Estuary (Figure 1), was constructed in 1994 to increase flushing between the Peel–Harvey Estuary and the ocean (Brearley 2005). A channel simply known as The Cut (Figure 2c) was opened in 1951 in the Leschenault Estuary, to increase drainage and reduce flooding (Semenuik et al. 2000). The opening of the channels has caused a marked increase in the influence of marine water in the lower reaches of the Serpentine, Murray and Collie Rivers, the three rivers chosen for this study.

De Lestang et al. (2003) plotted mean monthly salinities in the Peel Inlet, Harvey Estuary and Serpentine River for pre-Dawesville Cut (1980–1981) and post-Dawesville Cut (1995–1998) periods, and noted that during October to December mean monthly salinities were about 10–21 higher in the 1995–1998 interval compared to the 1980–1981 period. Their generalised data showed that after the Dawesville Cut was constructed mean monthly salinities in the main estuarine lagoon (Peel Inlet) varied from just above 40 in December–January to below 20 in August. The mouths of the Serpentine and Murray Rivers lie over 5 km across Peel Inlet from the natural open-marine inlet channel at Mandurah (Figure 1) and over 12 km across the estuarine lagoon from the artificial Dawesville Cut (Figure 2), the other open-marine inlet channel to the estuarine system. In contrast the mouth of the Collie River lies within 2 km of the open-marine channel to the Leschenault Inlet and is influenced by estuarine lagoon waters with an annual salinity variation of from about 29 at the end of winter to about 37 at the end of summer (Semeniuk et al. 2000).

Climate

The southwest of Western Australia is characterised by a temperate Mediterranean climate, with warm dry
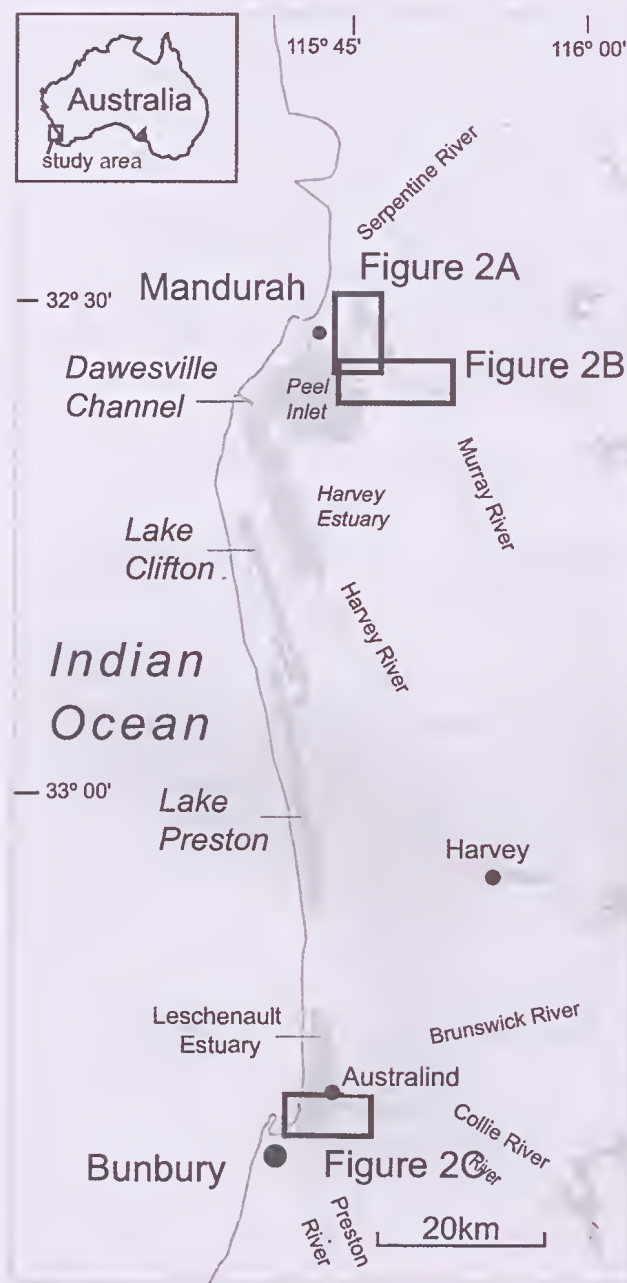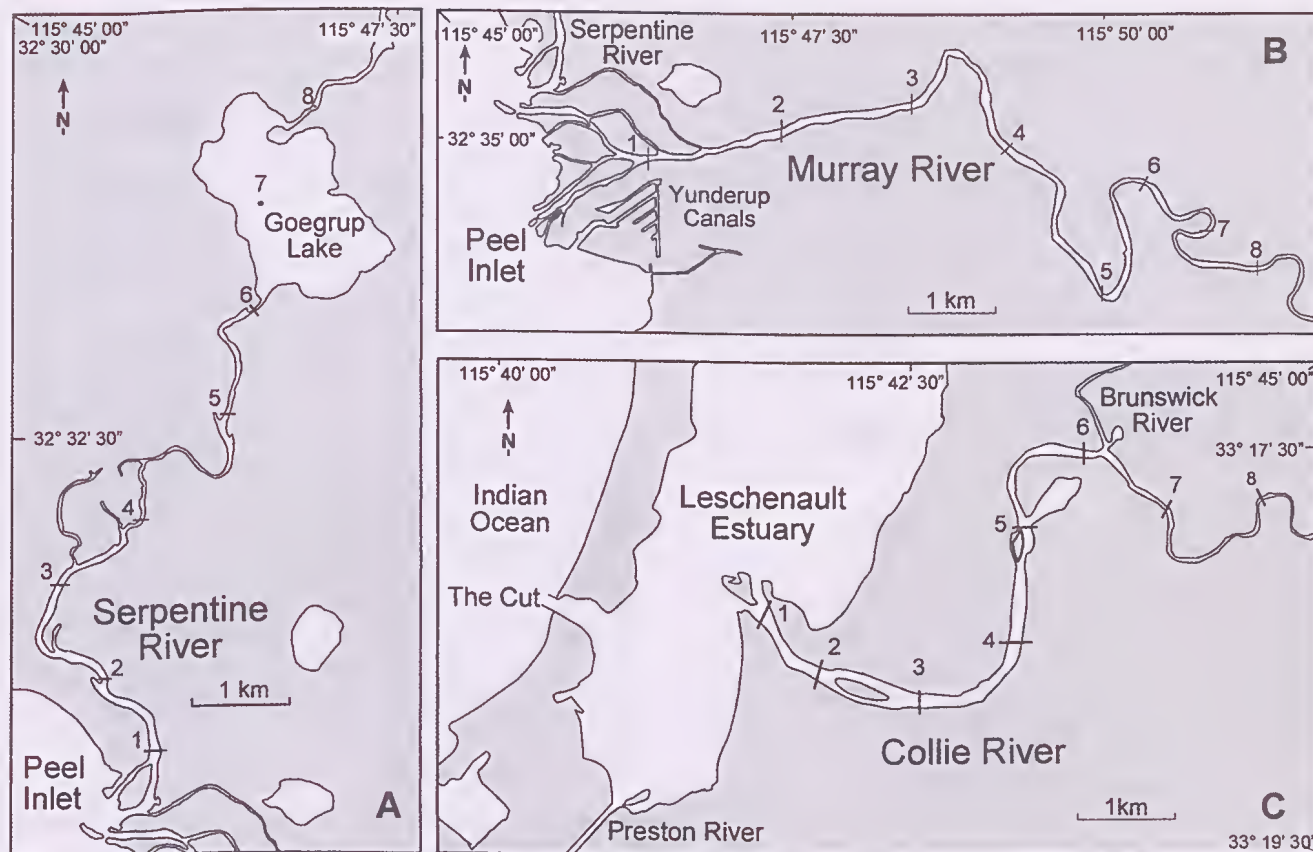
Figure 2 Location of studied transects in the Serpentine River (a), Murray River (b) and Collie River (c).

summers and cool wet winters. Average annual rainfall for Mandurah and Bunbury is 875 mm and 861 mm respectively. Rainfall is highly seasonal (Figure 3), with 90% of rainfall falling between May and October (Brearley 2005), though up to 95% of freshwater runoff has been recorded between June and September (C Wilson, J Hale & E I Paling unpubl. 1997 data). Mean annual evaporation is around 1750 mm, and similar to rainfall, is highly seasonal (C J Hearn, R J Lukatelich & J R Hunter unpubl. 1986 data).

The daily open ocean tidal range is around 0.6–0.7 m, and although there are no tide predictions for the Leschenault Estuary, in Peel Inlet the tidal range is around 0.4–0.5 m (www.bom.gov.au). The phase of the diurnal tide varies by 24 hours throughout the year, so that the ebb tide occurs during the night in summer, and during the day in winter (C J Hearn, R J Lukatelich & J R Hunter unpubl. 1986 data).

The mean maximum and minimum atmospheric temperatures experienced by Mandurah at the mouth of the Peel–Harvey Estuary are 29.5°C and 15.3°C in summer, and in winter, 17.3°C and 8.6°C. Summer maximum and minimum temperatures for Bunbury at the mouth of the Leschenault Estuary are 27.8°C and 13.9°C, and in winter 16.8°C and 8.4°C respectively (www.weatherbase.com).

Previous research

Published water-quality data on estuarine environments in southwest Western Australia is focused on the larger coastal estuaries, or on rivers that drain directly into the ocean (Brearley 2005; Semenuik et al. 2000). Comparatively little has been published on the tidal rivers that drain into the estuarine lagoons. The Serpentine, Murray and Collie Rivers and their associated estuaries were not included in the study of riverine sediment loads and the relationship between sediment and water quality in the wave-dominated estuaries of southwest Australia reported on by Radke et al. (2004).

Little has been published on the foraminiferal distributions in microtidal rivers in southwest Australia. The main studies have been performed on the large coastal estuaries (McKenzie 1962; Quilty 1977; Revets 2000; Quilty & Hosie 2006). Quilty's (1977) study on the Hardy Inlet, to the south of the present study area, and McKenzie's (1962) study of Oyster Harbour near Albany on the southern coast of Western Australia included a few localities in the lower reaches of tidal rivers associated with the estuaries. None of these investigations included an analysis of the influence of seasonal variation on the foraminiferal distributions in the tidal rivers or in the estuaries.

METHODS

Study sites

The Serpentine, Murray and Collie Rivers were sampled on three occasions, from 18–20 August 2005 (peak freshwater outflow, minimum tidal influence), 2–4 November 2005 (decreasing freshwater outflow) and the
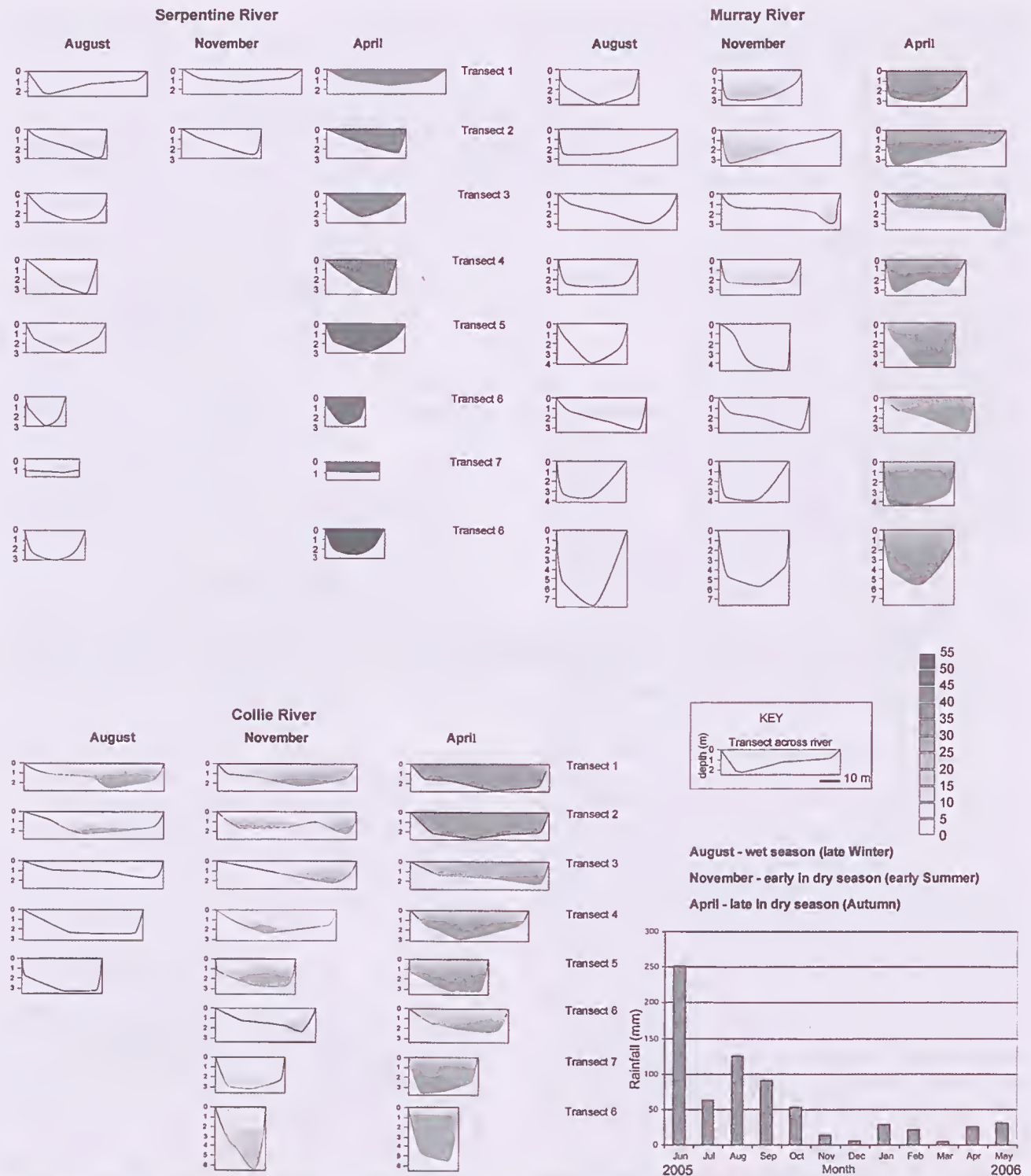
Figure 3 Salinity profiles for studied transects in each river during the August and November 2005 and April 2006 sampling periods. Also shown is a plot of average monthly rainfall from June 2005 to May 2006 (data from www.bom.gov.au).

5–7 April 2006 (negligible rainfall, maximum tidal influence) to gain an understanding of the annual variability in water quality. The rivers were not sampled in late January or February, as the summer of 2005–2006 was wetter, and the autumn drier, than long-term averages. Along about the lower 10 km of each river, eight transects (perpendicular to the river bank) were selected (Figure 2), and on each transect data on

bathymetry and water quality was obtained and a sediment sample retrieved, usually from three sites. Two of the sample sites were positioned 4–5 m from opposite sides of the river and one site was positioned at the centre of the river. Where transect distance was 60 m or greater, four sites were sampled with the central localities positioned about 20–25 m from each bank. Samples were numbered according to transect and a numerical series

starting from the left side of the river when facing upstream.

Salinity and sediment sampling

Because of the importance of salinity in controlling foraminiferal distributions (Murray 1991, 2006) and because salinity is one of the most variable features of the studied rivers, the relationship between salinity and the distribution of foraminiferal species is the focus of this study. Salinity was measured using a Hydrolab multiprobe at each sample site (Table 1*). Foraminifera live on or within the sediment forming the riverbed. Sediment was sampled using a sediment corer of 4 cm diameter, attached to a metal pole, operated from a boat. The top 2–3 cm of the recovered core was placed into a sample vial and stored in an icebox. Samples were stained with Rose Bengal one day after collection, and then wet sieved through 1 mm, 150 μm and 63 μm mesh. The sediment fractions were examined separately for foraminiferal analyses. In additional to salinity, water temperature and pH were measured as well as dissolved oxygen for some samples (Table 1*). Grainsize analysis was also undertaken using a Sedigraph X-ray attenuation instrument for the fraction less than 3.25 Φ (106 μm), and using a settling tube for the >106 μm fraction (Table 2*).

Foraminiferal analysis

All sediment samples were first examined to determine which species were present. The 1 mm–150 μm fraction for the most diverse sample in each transect collected during August and April was then systematically picked. For each of these samples, an attempt was made to pick 200 specimens in a total pick (dead and live individuals) and 100 in a pick of live individuals (identified by Rose Bengal staining), but for many of the samples this was not possible particularly for the live assemblage. As well as the total systematic pick, where possible an additional search for very rare species was made. Live foraminifera, identified by Rose Bengal staining, were extremely rare and in all samples the total pick approximates the dead assemblage. Due to the difficulty in identifying living agglutinated species with Rose Bengal the analysis of the live microfauna is limited to hyaline and porcelaneous species.

For the August samples, species were classified as being present or absent, and clustered using the Jaccard Index and the UPGMA method in MVSP 3.1. The statistical foraminiferal data was clustered using the Bray–Curtis method, with data square-root transformed, as clustering using the Bray–Curtis method with unweighted data has been recognised as producing clusters influenced by high values (Stephenson 1973; Michie 1982). Foraminifera and washed-sediment residues recovered in this study are in the collections of the Edward de Courcy Clarke Earth Science Museum at the University of Western Australia.

RESULTS

Salinity

Due to variation in rainfall between wet winters and dry summers significant seasonal changes in salinity take place in each river (Table 1*). Salinity profiles (Figure 3) observed at each transect early in the dry season (November 2005), late in the dry season (autumn 2006), and during the wet season (August 2006) illustrate major differences between the rivers. Salinity stratification was most apparent in the Collie River at all measuring times, and in the records taken from the Murray River during November and April. The highest salinities were found throughout the Serpentine River during the April sampling.

In the bottom waters of all the rivers (Figure 4), except near the mouth of the Collie River, salinities lower than 2 are present during the wet season and salinities higher than 30 prevail at the end of the dry season. The maximum variation between wet- and dry-season bottom-water salinities of slightly over 50 was found in
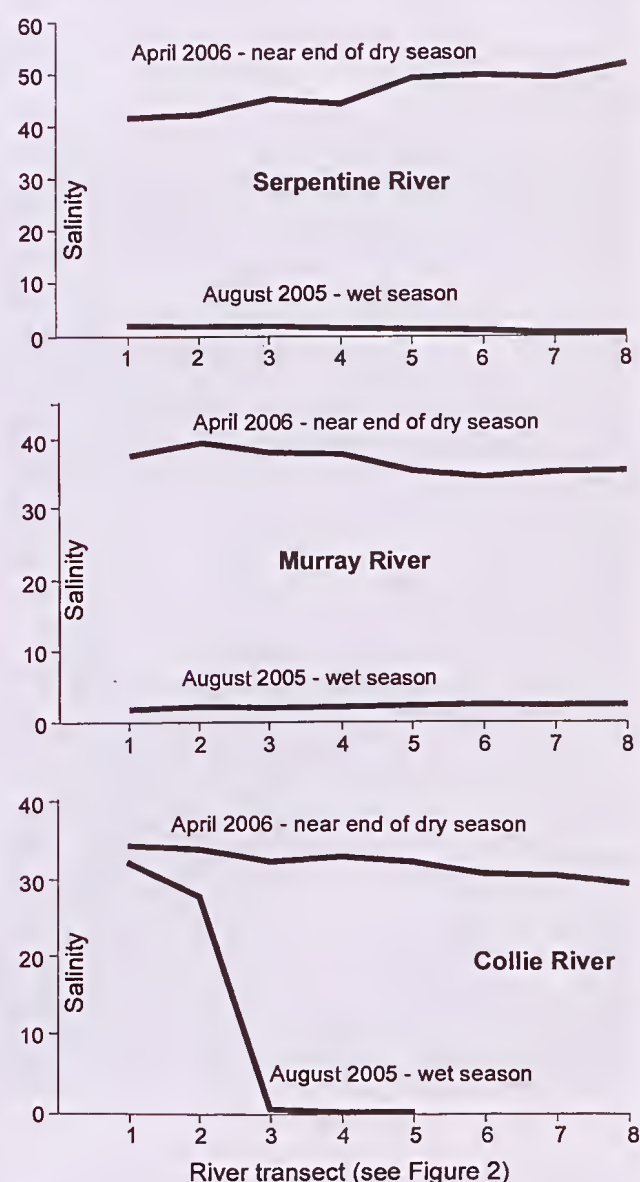

Figure 4 Comparison of bottom-water salinity at each river transect between the August 2005 wet season and the April 2006 end of dry season. The salinity measurements are those from the deepest part of the river sampled on each transect.

transect 8 of the Serpentine River. During the dry season, bottom-water salinities in the Murray and Collie Rivers decrease slightly upstream, whereas those in the Serpentine River show a slight increase upstream.

The reasons for the salinity differences between rivers are probably: (i) the broader estuarine lagoon of lower winter salinity into which the Serpentine and Murray Rivers flow compared to the narrow southern part of the Leschenault Estuary (of almost normal-marine salinity in winter) into which the Collie River flows; (ii) the smaller freshwater catchment area and therefore lower freshwater outflow in the Collie River compared to the other rivers; and (iii) the deeper less-restricted river channel in the Murray River compared to the Serpentine River; and the broad very shallow Goegrup Lake on the Serpentine River (Figure 2a) that because of evaporation may be responsible for elevated salinities here during the dry season.

Foraminiferal distributions

Thirty-five foraminiferal species have been found in the studied rivers (Figures 5, 6; Appendix 1). These include types well known from estuaries and tidal rivers elsewhere in Australia and on other continents. As demonstrated by Hayward *et al.* (2004) in their study of *Ammonia*, the taxonomy of many of the species, even those most widely recorded, is poorly understood. For this reason we have used broad species groups for some taxa (e.g. *Ammonia* gr. *tepida*) rather than more precise determinations.

LIVING (STAINED) ASSEMBLAGES

Only hyaline and porcelaneous species are considered here because of the difficulty in recognising stained individuals among the agglutinated foraminifera. The total (living plus dead) distribution of foraminiferal tests (see below) suggests that organic-cemented agglutinated foraminifera live through a broad salinity range in the rivers. Tables 3*–5* record the August (wet season) and April (dry season) fauna living in each river and Figure 7 summarises these distributions against salinity. The results presented here are preliminary and based on low stained-specimen counts and on the assumption that tests
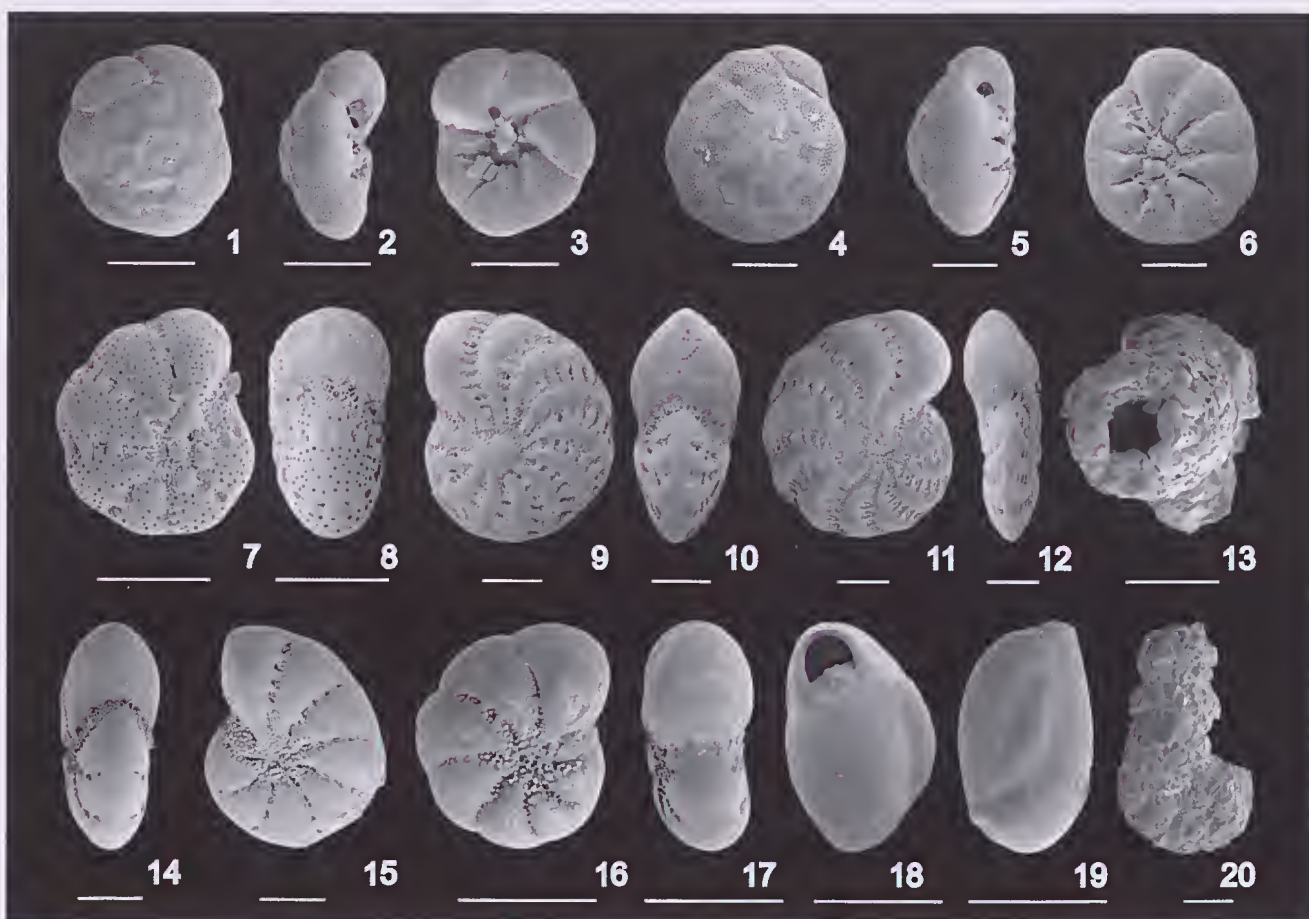

Figure 5 Dominant foraminiferal species; secondary electron images. Scale bars are 0.1 mm. (1–3) *Ammonia* gr. *tepida* (Cushman), from Murray River: (1) spiral view, (2) peripheral view, (3) umbilical view. (4–6) *Ammonia* gr. *tepida* (Cushman), from Murray River: (4) spiral view, (5) peripheral view, (6) umbilical view. (7, 8) *Elphidium* gr. *gunteri* Cole, from Murray River: (7) lateral view, (8) peripheral view. (9, 10) *Elphidium advenum* (Cushman), from Collie River: (9) lateral view, (10) peripheral (apertural) view. (11, 12) *Elphidium* cf. *advenum* (Cushman), from Collie River: (11) lateral view, (12) peripheral view. (13, 20) *Ammobaculites exiguus* Cushman & Bronnimann, from Serpentine River: (13) apertural view, (20) lateral view. (14, 15) *Elphidium excavatum* (Terquem), from Collie River: (14) peripheral view, (15) lateral view. (16, 17) *Elphidium* cf. *excavatum* (Terquem), from Murray River: (16) peripheral view, (17) lateral view. (18, 19) *Quinqueloculina* gr. *seminula* (Linnaeus), from Collie River: (18) apertural view, (19) lateral view.

Figure 6 Rare foraminiferal species not previously illustrated from estuaries in southwest Australia; secondary electron images. All specimens from the Collie River. Scale bars are 0.1 mm. (1–3) *Aubignyna* sp.: (1) spiral view, (2) umbilical view, (3) peripheral view. (4) *Bulimina marginata* d'Orbigny: lateral view. (5) *Buliminella elegantissima* d'Orbigny: lateral view. (6, 7) *Neoconorbina* sp.: (6) spiral view, (7) umbilical view. (8–10) *Paratrochammina* sp.: (8) spiral view, (9) umbilical view, (10) peripheral view. (11–13) *Rosalina* sp.: (11) spiral view, (12) peripheral view, (13) umbilical view. (14, 15) *Spiroloculina* sp.: (14) lateral view, (15) peripheral view. (16–18) *Trochammina?* sp.: (16) umbilical view, (17) peripheral view, (18) spiral view.

showing internal red staining represent live individuals. In future studies, examination of larger sediment samples may yield greater specimen counts that may be statistically treated; and a combination of CellTracker Green and Rose Bengal staining viewed under epifluorescence microscopy (Bernhard *et al.* 2006) may provide better detection of living individuals.

In the Serpentine River, the assemblage during August (wet season) was dominated by *Ammonia* gr. *tepida*. *Elphidium* gr. *gunteri* and *Elphidium* gr. *excavatum* were the only other species groups living in the river. The numbers of living foraminifera were sparse, and populations of individual species were highest toward the river mouth. In contrast to the wet season pattern, six of seven hyaline and porcelaneous species in the overall (dead + living) assemblage were living in April at the end of the dry season. The diversity of the living assemblage steadily decreased from transects 1 to 5 and *Ammonia* gr. *tepida* was the only species that was living upstream from transect 5.

The distribution of foraminifera living in the Murray River in August was similar to that in the Serpentine River. *Ammonia* gr. *tepida* was the most common living species group, and *Elphidium* gr. *gunteri* was living mainly toward the river mouth. Living individuals decreased in abundance upstream, although the trend was not as noticeable as in the Serpentine River. Three species groups, *Ammonia* gr. *tepida*, *Elphidium* gr. *gunteri* and *Quinqueloculina* gr. *seminula* were living at more than one location in the Murray River during April. Two of these groups, *A.* gr. *tepida* and *E.* gr. *gunteri* were living throughout the river during this month, whereas *Q.* gr. *seminula* was living mainly in transects 1 and 2. *Cornuspira planorbis* was living in one sample, and *Elphidium excavatum* subsp. 2 was living in one sample and inconclusively stained in two others.

The highest number of living species during the wet season (August) was in the Collie River. Although a total of nine different species (and possibly two others) were stained, only three species groups, *Quinqueloculina* gr.
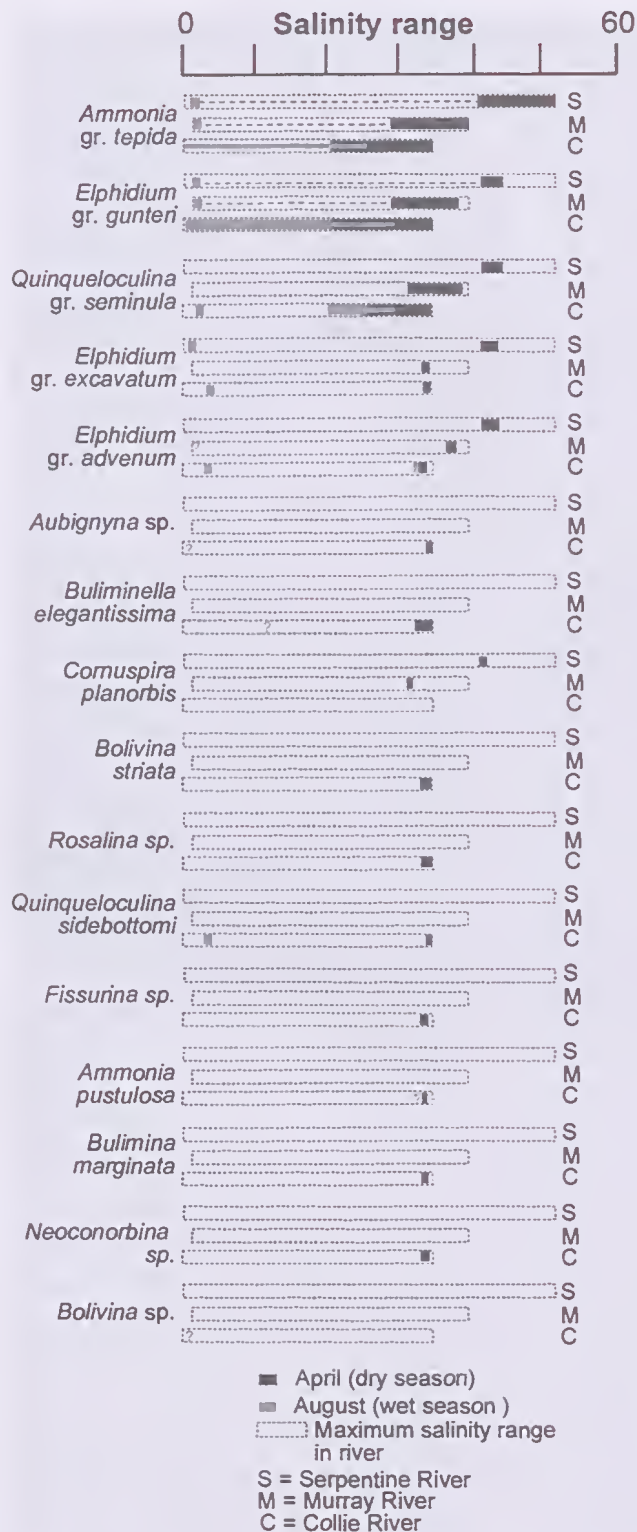
Figure 7 Summary of the distributions of living (Rose-Bengal stained) foraminifera plotted against salinity. Distributions in each river are recorded in Tables 3*–5*.

seminula, *Ammonia* gr. *tepida* and *Elphidium* gr. *gunteri*, were living at more than one location. There was a marked decrease in living species upstream. Of the 24 hyaline and porcelaneous species found in the Collie River as dead or alive specimens, 16 were living in April

(end of dry season). The majority of tests were living within the first two transects, or in the deeper sites of transects 3 and 4. As in the Murray and Serpentine Rivers, living specimens of *Ammonia* gr. *tepida* were present in each transect examined in the Collie River.

The decrease in diversity of foraminifera upstream from the mouth of each river is attributed to a decrease in salinity, as is the lower diversity found in the August wet season compared to the April dry season (Figure 7). The much greater diversity in the lower three transects (1–3) of the Collie River compared to the Serpentine and Murray River transects, is coincident with a greater marine influence in the Collie River throughout the year and the more stable bottom-water salinity conditions in transects 1–3 of this river. Of the hyaline species found in the rivers, *Ammonia* gr *tepida* is most tolerant to salinity variation, occupying the full salinity range available to it in the studied sections of each river. *Elphidium* gr. *gunteri*, *E.* gr. *excavatum* and *E.* gr. *advenum*, while not as abundant as *Ammonia* gr. *tepida* and less tolerant in the upper salinity range (>44), also have broad salinity ranges. Among the porcelaneous species, small thin-walled *Quinqueloculina* gr. *seminula* has the most extensive distribution in the rivers (mainly 24–45). The records of rare *Q.* gr. *seminula* and *Q. sidebottomii* at low salinities in transects 1 and 2 of the Collie River may be of living tests slightly displaced from more saline waters of the salt wedge in this section of the river.

TOTAL (DEAD PLUS LIVING) ASSEMBLAGES

A total of 10 foraminiferal species including three agglutinated, two porcelaneous and five hyaline species, were identified from the Serpentine River among the total assemblage (Figure 8) based on samples collected during August sampling. Living individuals form a very small component of the total assemblage. *Ammonia* gr. *tepida* dominates the total assemblage and *Ammobaculites exiguus* is the subdominant species. *Quinqueloculina* gr. *seminula* occurs with a frequency of about 10% in the total foraminiferal assemblage in sample S2.1, but is rare elsewhere in transects 1–6. All other species are present at frequencies less than 10% of the total assemblage in any one sample.

Based on presence and absence distribution of species, the samples collected from the Serpentine River cluster in two main groups, one present in transects 1–5 and the other recorded from transects 5–8 (Figure 8). Species diversity decreases upstream from nine species present in the transect 1–5 cluster, and four species present in the transect 5–8 cluster. *Ammobaculites exiguus*, *Ammonia* gr. *tepida*, *Elphidium* gr. *gunteri*, *Elphidium* gr. *excavatum* and *Quinqueloculina* gr. *seminula* each occur in at least 12 of the 13 samples making the transect 1–5 cluster. Porcelaneous species are present only in this cluster, with *Cornuspira planorbis* and *Quinqueloculina* gr. *seminula* most abundant closest to the river mouth. *Quinqueloculina* gr. *seminula* is very rare above transect 2 where it is represented by very small fragile tests. *Ammobaculites exiguus* and *Ammonia* gr. *tepida* are present in each sample of the transect 5–8 cluster in the upstream reach of the tidal river, and these are associated with sporadic *Elphidium* gr. *gunteri* and *Trochammina inflata*.
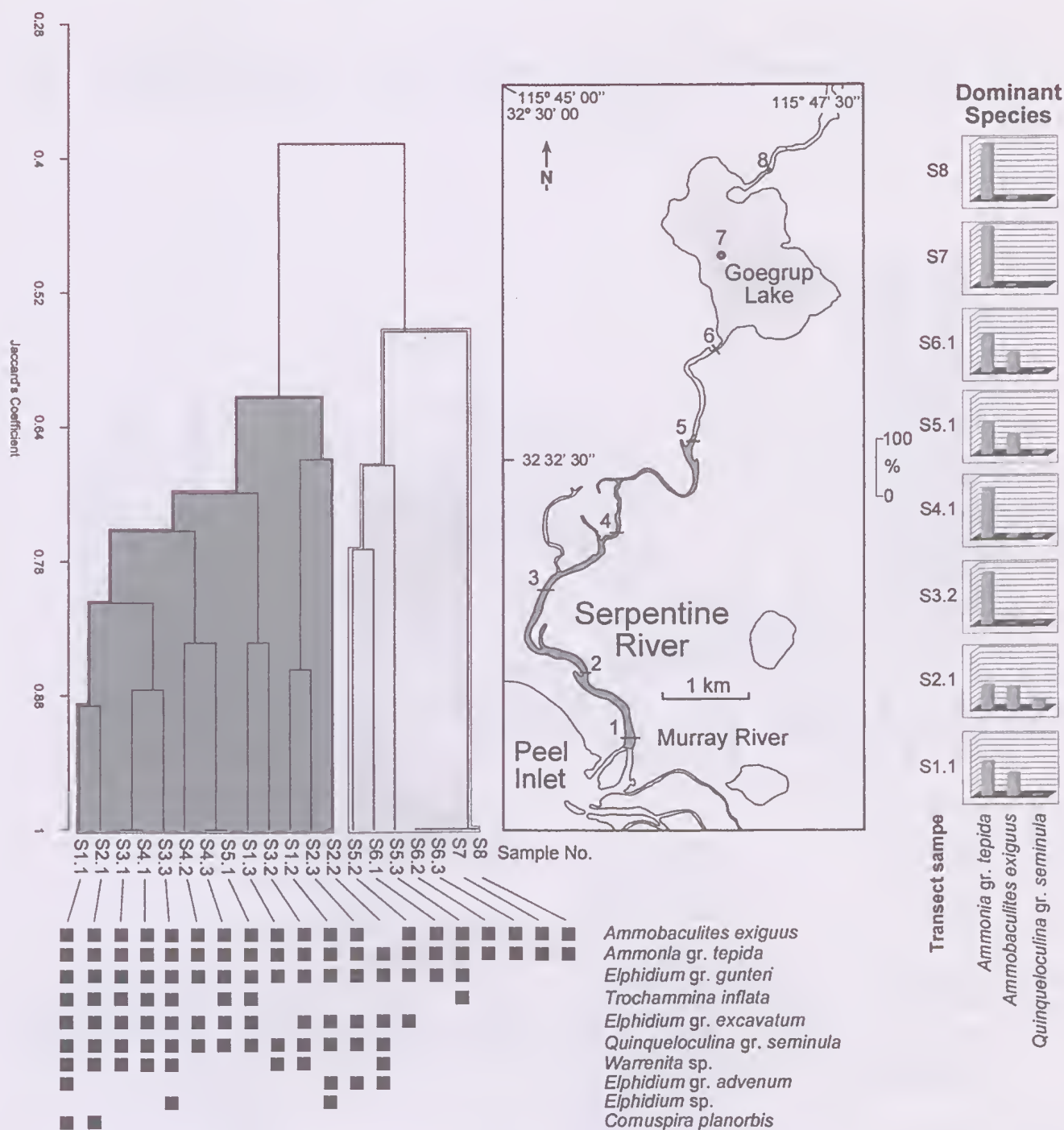
Figure 8 Distribution of foraminifera (total assemblage) from the Serpentine River during August 2005. Dominant species are those that have a frequency of ≥10% in one or more samples. The cluster analysis is based on the Jaccard Index and the UPGMA method in MVSP version 3.1.

In the Murray River the first transect undertaken during August sampling was located where the inlet channel had been dredged. There were almost no foraminifera in the sample from the dredged area, which was composed of compacted clay, and large proportions of reworked foraminifera were present in other samples in the transect. As these samples are not representative of modern conditions, sampling was moved about 300 m downstream in April (as shown in Figure 9). Samples were clustered using the samples collected in April.

A total of 14 species, five agglutinated, seven hyaline and two porcelaneous, were identified from the Murray River in the total foraminiferal assemblage (Figure 9). As in the Serpentine River, *Ammonia* gr. *tepida* is the dominant species, but *Elphidium* gr. *gunteri* is the only other species present at frequencies above 10% in at least one sample. In contrast to the Serpentine River, *Ammobaculites exiguus* was the least common agglutinated foraminifera in the Murray River. The most common agglutinated species is *Warrenita* sp.
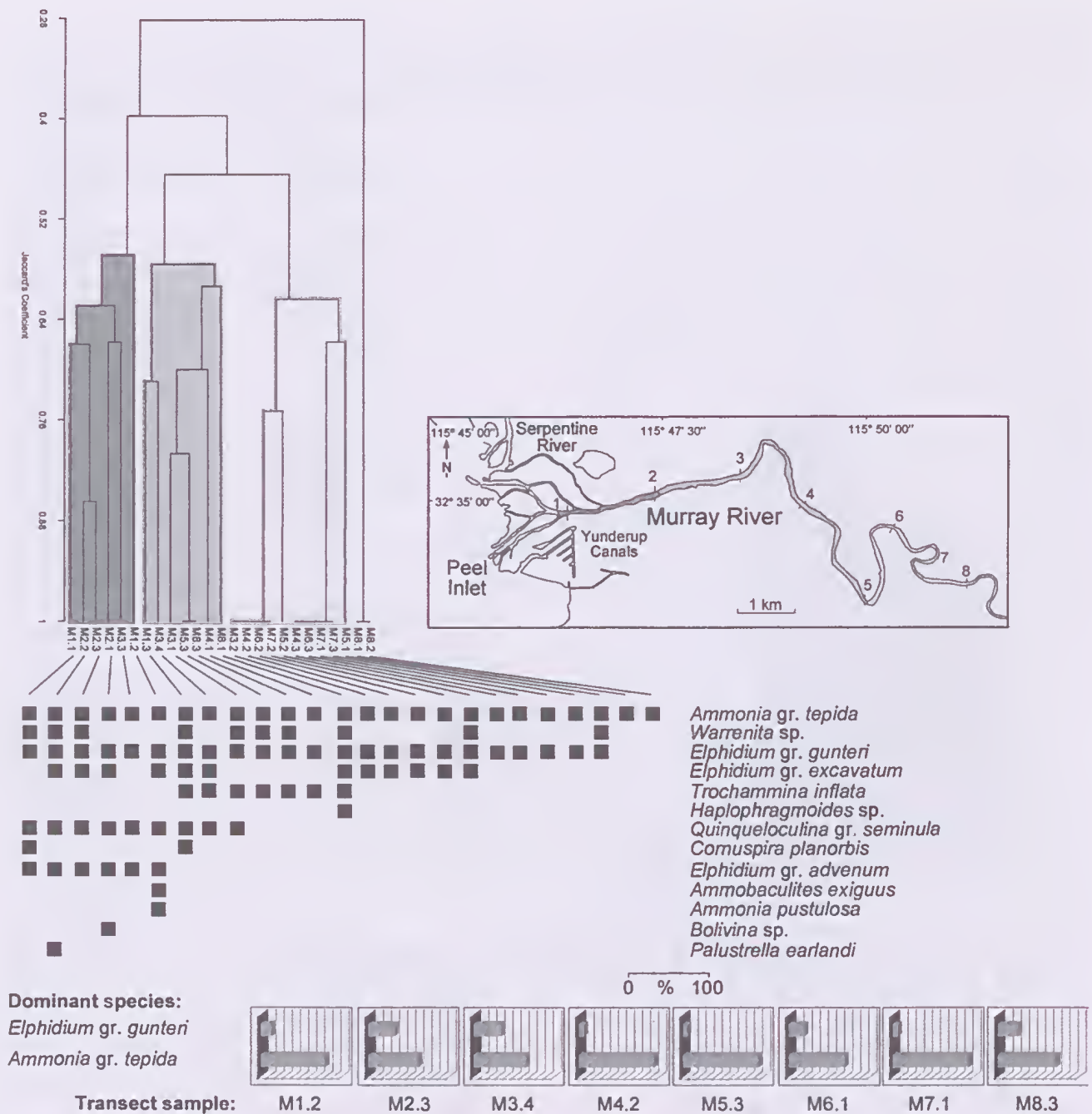
Figure 9 Distribution of foraminifera (total assemblage) from the Murray River during April 2006. Dominant species are those that have a frequency of ≥10% in one or more samples. The cluster analysis is based on the Jaccard Index and the UPGMA method in MVSP version 3.1.

Porcelaneous species (*Quinqueloculina* gr. *seminula* and *Cornuspira planorbis*) are present in transects close to the river mouth.

Based on presence–absence species distributions, three sample clusters were defined (Figure 9). These loosely distinguish upper, middle and lower reaches of the tidal river based on a decrease in species diversity upstream rather than on discrete suites of species. The lower cluster is made up of samples in transects 1–3. Most samples forming the middle cluster are from transects 3–6, with one sample from transect 1 and one from transect 8.

Species diversity in some samples in the middle estuary section were lower than expected, and are clustered in the upstream cluster which includes samples from transects 3–8. This cluster pattern seems to be due to sediment grainsize because where samples were taken from well-sorted coarse sediment (e.g. samples M3.2, M4.2, M4.3, M5.2: Table 2*), species diversity and abundance are much lower than expected, and often *Ammonia* gr. *tepida* is the only species group present. Another anomaly in the cluster pattern is the presence of sample 8.3 in the middle cluster. In transect 8, sites M8.1
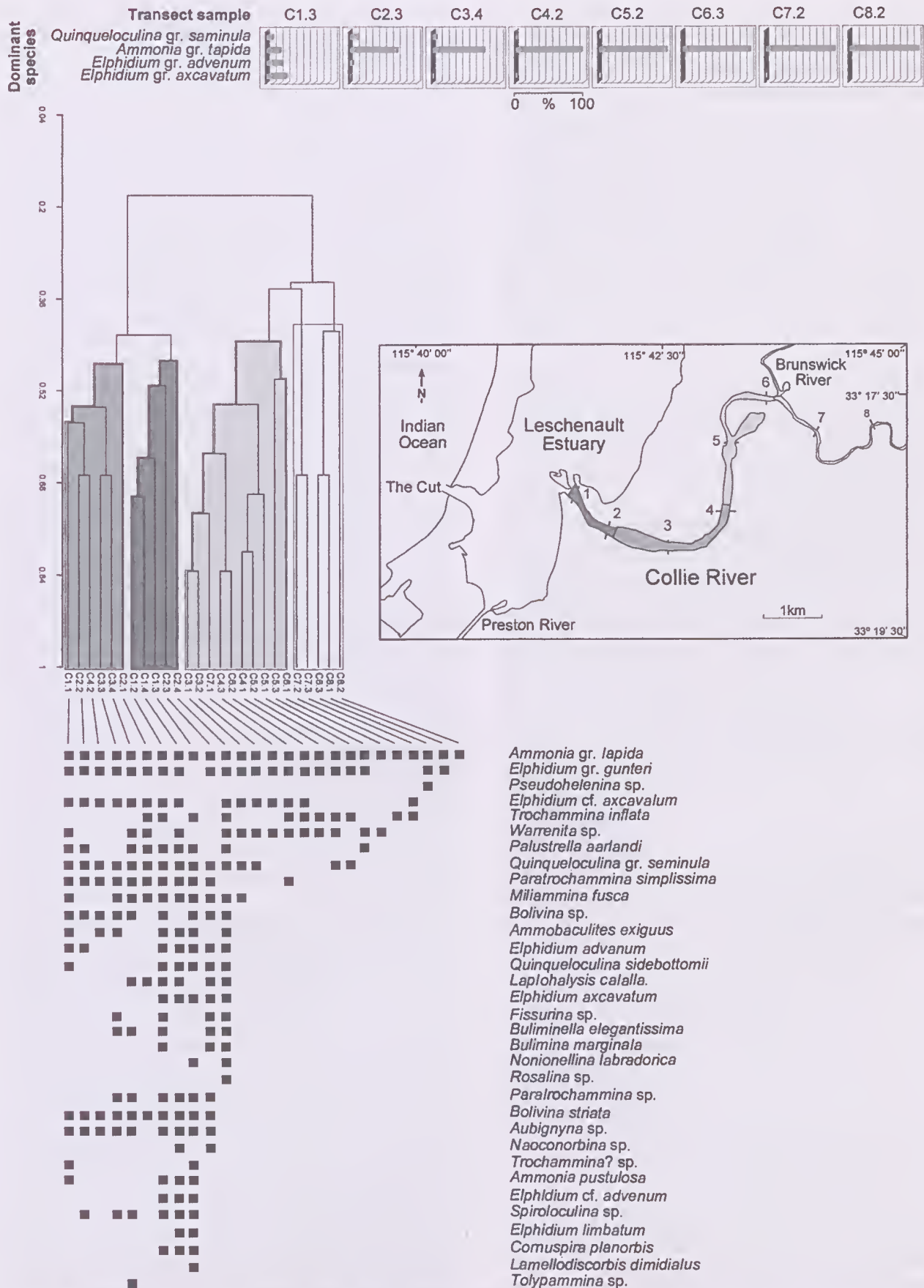
Figure 10 Distribution of foraminifera (total assemblage) from the Collie River during August 2005. Dominant species are those that have a frequency of ≥10% in one or more samples. The cluster analysis is based on the Jaccard Index and the UPGMA method in MVSP version 3.1.

and M8.2 were influenced by bottom-water anoxia during the April sampling period, whereas site M8.3 was well oxygenated which may explain its greater foraminiferal diversity compared to other samples in the upper tidal reaches of the river.

The most diverse foraminiferal microfauna found in any of the rivers was in the lower reaches of the Collie River, with 33 species identified, nine agglutinated, four porcelaneous and 20 hyaline (Figure 10). Four species groups have frequencies greater than 10% of the total foraminiferal assemblage in at least one of the studied sites in the river (Figure 10). Of these, *Ammonia* gr. *tepida* is the only species present in each transect; *Quinqueloculina* gr. *seminula* is present in transects 1–4; *Elphidium* gr. *advenum* is confined to transects 1–4; and *Elphidium* gr. *excavatum* has a broader range in transects 1–7.

Based on presence–absence foraminiferal distributions, four clusters of samples are distinguished in the Collie River (Figure 10), defined by decreasing diversity upstream. The most diverse cluster, with 31 species, includes samples from transects 1 and 2 at the mouth of the river. Species confined to the cluster include hyaline *Bulimina marginata*, *Cornuspira planorbis*, *Elphidium* cf. *advenum*, *Elphidium limbatum*, *Elphidium excavatum*, *Lamellodiscorbis dimidiatus*, *Neoconorbina* sp., *Nonionellina labradorica* and *Rosalina* sp. A second cluster which includes samples from transects 1–4 has 23 species including agglutinated *Tolypammina* sp. found only in this cluster at a single locality, and agglutinated *Ammobaculites exiguus*, *Leptohalysis catella*, *Paratrochammina* sp. and *Trochammina?* sp., porcelaneous *Quinqueloculina sidebottomii* and *Spiroloculina* sp., and hyaline *Ammonia pustulosa*, *Aubignyna* sp., *Buliminella elegantissima*, *Elphidium advenum*, *Fissurina* sp. and *Bolivina striata* shared with the first cluster described above. A third cluster includes samples from transects 3–7 and is characterised by much lower diversity, with eight species present, than the clusters downstream. The most persistent group in this cluster is hyaline *Ammonia* gr. *tepida* present in each sample. Hyaline *Elphidium* gr. *gunteri* and agglutinated *Warrenita* sp. are present in nine of the 10 samples. Porcelaneous *Quinqueloculina* gr. *seminula* has a sporadic occurrence in samples of the cluster, whereas it is present in each sample of the downstream clusters. Other species sporadically occurring here but more common downstream include agglutinated *Miliammina fusca*, *Palustrella earlandi* and *Paratrochammina simplissima*. Hyaline *Elphidium* cf. *excavatum* and agglutinated *Trochammina inflata* are more persistent and occur both in clusters downstream and upstream. The least diverse cluster includes samples from transects 6–8 in the upstream reaches of the tidal river. Here five species are present including *Ammonia* gr. *tepida* which is present in all samples. Hyaline *Pseudohelenina* sp. is found only in this cluster.

The influence of bathymetry on salt-wedge distribution, and variations in foraminiferal assemblage composition within a single transect are most evident in the Collie River compared to the other rivers. Upstream of transect 2 in the Collie River, the majority of species are confined to the deepest samples in transects C3 and C4. In transect 3, seven species are identified in samples C3.1 or C3.2, whereas 18 species identified in samples

C3.3 or C3.4. Sample C4.2, the deepest site along transect 4, although located further upstream, has a similar diversity to samples C3.1 or C3.2.

DISCUSSION

Interpretation of our results is qualified by: (i) an assumption that the Rose-Bengal stained specimens represent living foraminifera; (ii) the very low abundance of stained foraminifera that precludes statistical analysis of their distribution patterns; and (iii) a lack of data on sediment mobility within the rivers. During the dry season, because of microtidal conditions and lack of freshwater outflow, there may be little sediment transport in the rivers. However in the wet season, the increased freshwater outflow may lead to down-river transport of sediment, including foraminiferal tests. The results we present should be considered preliminary until more detailed investigations can be made. However, our results suggest that similar trends in foraminiferal distribution patterns are present in each river and we make some tentative conclusions about these patterns.

Faunal diversity is most probably influenced by the amount of variation in salinity that occurs during the year and the residence times of normal-marine salinity (as against hyposaline and hypersaline conditions) during the dry season. The Collie River with the least variation in salinity (up to 32) has the most diverse assemblage at least in the lower reaches of the river where bottom waters maintain stable normal-marine conditions throughout the year. Some normal-marine species have apparently been able to migrate into the Collie River from the Leschenault Inlet without encountering any substantial low-salinity barrier throughout the year. The Serpentine River with the most variation in salinity (up to 51) has the least diverse assemblage. Species with low tolerance to variable salinity apparently have had difficulty migrating into this river (and the adjacent Murray River) because of the low salinity during winter in Peel Inlet and the absence of a salt wedge in the lower reaches of the rivers during this season.

The distribution of *Ammonia* gr. *tepida*, present among the total assemblage at each site in each river, suggests that it is the species most tolerant of salinity variation. During the wet season, live *A.* gr. *tepida* was found in very low salinities in the Serpentine River upstream to transect 5, in the Murray River upstream to transect 8, and in the Collie River upstream to the highest transect sampled during August. Similarly live *Elphidium* gr. *gunteri*, although more sporadic than *A.* gr. *tepida*, was found to have a broad distribution upstream during the wet season. In the Serpentine and Murray Rivers, other species were only found live during the dry season, and presumably are transported into the river with an advancing salt wedge. The sharp decrease in diversity in transects 3 and 4 of the Collie River compared to transects 1 and 2, suggests that the upper tidal reaches of this river experience similar conditions to the tidal reaches of the Serpentine and Murray Rivers, and this is evidenced by similarities in the faunal assemblages.

Based on the distributions of the live hyaline and porcelaneous foraminifera in all rivers (Figure 7), the

order of tolerance to salinity changes seems to be: *Ammonia* gr. *tepida* with highest tolerance; *Elphidium* gr. *gunteri*, *E.* gr. *excavatum* and *E.* gr. *advenum* with slightly lower tolerance; *Quinqueloculina* gr. *seminula* with moderate tolerance; and *Cornuspira planorbis*, *Bolivina striata*, *Rosalina* sp. *Aubignyna* sp., *Bolivina* sp., *Buliminella elegantissima*, *Fissurina* sp., *Bulimina marginata*, *Quinqueloculina sidebottomii*, *Neoconorbina* sp. and *Ammonia pustulosa* with low tolerance. Among the organic-cemented agglutinated foraminifera, based on total assemblages, the distribution of *Ammobaculites exiguus* in the Serpentine River suggests that it has high tolerance to salinity changes. However, this species was rare in the nearby Murray River (found at one site in the lower reaches) and was absent from the Collie River. The substrate type and the organic content of the sediment do not seem to control the distribution of *Ammobaculites* here, and some unknown factor must be involved. *Trochammina inflata*, *Warrenita* sp. probably have moderate tolerance to salinity changes; whereas *Palustrella earlandi*, *Paratrochammina simplissima*, *Miliammina fusca*, *Leptohalysis catella*, *Paratrochammina* sp., *Trochammina*? sp. and *Tolypammina* sp. probably have lower tolerance. Calcite-cemented agglutinated foraminifera are absent from the rivers in contrast to their dominance in open-marine agglutinated assemblages along the southwest Australian coast (Haig 2002).

A detailed biogeographic analysis of the fauna is beyond the scope of this paper and awaits more comprehensive studies of other estuaries in the region. However, some preliminary biogeographic observations can be made. In southwest Australia, foraminiferal assemblages comparable to those found in the studied rivers were recorded by Quilty (1977) in the tidal reaches of the Blackwood–Scott river system leading into Hardy Inlet, and from the upper tidal reaches of the Swan River (Corr 1998). Studies of seasonal variation in the foraminiferal distributions have not been made in these rivers. Like the Collie River, the Blackwood–Scott river system has seasonal salinity variation of up to about 30 and the rivers show a similar rapid decrease in foraminiferal diversity upstream from the main inlet, from a moderately high number of species in the inlet (47 in Hardy Inlet) to assemblages with dominant *Ammonia* gr. *tepida* and very few other species in the upper tidal river system. In the upper tidal reaches of the Swan River, Corr (1998) also recorded *Ammonia* gr. *tepida* as the dominant species group, with *Elphidium* gr. *gunteri* and *Elphidium* gr. *excavatum* present, but in much lower proportions. Sediment in the Swan Estuary in the lower reaches of the river has a much more diverse foraminiferal assemblage (Quilty & Hosie 2006) than that found in any sample from the Serpentine, Murray and Collie Rivers but the total assemblage described there may include a large component derived from mid-Holocene highstand (normal-marine) deposits that outcrop around the estuary (Yassini & Kendrick 1988).

Yassini & Jones (1995) reviewed southeast Australian estuaries, coastal lagoons and tidal rivers. In contrast to the Mediterranean climate of southwest Australia, southeast Australia has a warm temperate climate with hot wet summers and cool to mild wet winters. The seasonal salinity variation in estuaries and tidal rivers is therefore not as large as recorded in the southwest tidal

rivers and estuaries. Although assemblage distributions comparable to those found in the Serpentine, Murray and Collie Rivers are not obvious in the eastern Australian rivers there are some interesting similarities and differences in species distributions. For example, Yassini & Jones (1995) showed that *Ammonia* gr. *tepida* (recorded as *A. beccarii*) and *Ammobaculites exiguus* (as *Ammobaculites agglutinans*) have widespread distributions throughout estuarine environments. *Elphidium* gr. *gunteri* (as *Cribroelphidium vadescens*), is limited to coastal lagoon environments, and *Leptohalysis* sp., *Miliammina fusca* and *Trochammina inflata* also inhabit coastal lagoons. *Fissurina* sp. (=*F. lucida*), *Rosalina* sp., *Spiroloculina* spp. and *Bolivina* sp. prefer marine environments, and in brackish environments will only inhabit inlet channels. *Quinqueloculina* gr. *seminula* is the most widespread porcelaneous species in the brackish environments. These distribution patterns broadly mirror those found in southwest Australia, except that *Elphidium* gr. *gunteri* seems to have a more confined occurrence in the southeastern Australian estuaries and tidal rivers.

A comparison of foraminifera from brackish environments from southwest Western Australia and from other continents (Murray 1991, 2006) shows that a typical brackish-water assemblage is present in the Serpentine, Murray and Collie Rivers. There is, however, one unusual pattern in the foraminiferal distributions found in the studied rivers of southwest Australia. The upper regions of estuarine environments elsewhere are often dominated by organic-cemented agglutinated species (Murray 1991; Hayward *et al.* 1999; Scott *et al.* 2001; Debenay 2002). Although *Ammobaculites exiguus* is common in the Serpentine River, this pattern is not apparent in the Murray and Collie Rivers. Not only are there few organic-cemented agglutinated foraminifera throughout the latter rivers, but when the agglutinated species are present, their abundances are very low.

In a review of foraminifera from tropical paralic environments, Debenay (2000) showed approximate salinity ranges for *Ammonia* gr. *tepida* (5–80), *Elphidium* gr. *gunteri* (10–80), *Ammobaculites exiguus* (20–70), *Quinqueloculina* gr. *seminula* (10–60), that parallel, in order of salinity tolerance, the ranges found in southwest Australian rivers. The ranges recorded by Debenay (2002) are much broader than those found here, and the agglutinated species (*Ammotium salsum*, 1–105; *Haplophragmoides wilberti*, 2–100; *Arenoparrella Mexicana*, 1–100) with tolerances higher than the range of *Ammonia* gr. *tepida* have not been found in southwest Australia.

CONCLUSIONS

Three microtidal rivers studied in southwest Australia show large changes in salinity from the winter wet season to the summer dry season. The largest change, measured from August 2005 (during peak freshwater outflow in winter) to April 2006 (negligible freshwater outflow at end of dry season) was found to be 51 in the Serpentine River. The Murray River showed a maximum change of 38, whereas in the Collie River the maximum change was 32. The Collie River was the only river with a salt wedge of normal salinity at its mouth (in the deepest part of its central channel) during winter. Estuarine lagoon waters between the Collie River mouth

and the open-sea channel into the Leschenault Inlet remain within or only slightly below normal-marine salinity levels throughout the year. In contrast the Serpentine and Murray Rivers have well-mixed waters of very low salinity at their mouths during winter and open into the Peel Inlet, a broad shallow estuarine lagoon with winter salinities lower than 20. The seasonal changes in salinity in the tidal rivers and the adjacent estuarine lagoons and the changes in salinity upstream from the mouth of the river have apparently influenced the foraminiferal distributions. Preliminary findings from the present study are:

(1) Thirty-five foraminiferal species are recorded from the rivers including 11 organic-cemented agglutinated types, four porcelaneous species, and 20 hyaline forms. The majority of species are known from estuaries elsewhere. Thirty-three of the species are recorded from the Collie River, 14 from the Murray River, and 10 species from the Serpentine River. The agglutinated *Ammobaculites exiguus*, the porcelaneous *Quinqueloculina* gr. *seminula* and the hyaline *Ammonia* gr. *tepida*, *Elphidium* gr. *gunteri*, *E.* gr. *advenum* and *E.* gr. *excavatum* dominate total (dead + live) assemblages in various parts of the rivers.

(2) Only the Serpentine River shows a typical estuarine distribution pattern with organic-cemented agglutinated species common in the middle to upper reaches of the river; the other rivers have sparse agglutinated assemblages.

(3) Using total assemblages and cluster analysis, equivalent differentiation of faunas is recognised in each river with a consistent change in diversity upstream from the river mouth. The diversity of live hyaline and porcelaneous foraminifera (recognised from Rose-Bengal stained specimens) shows a similar decrease to that of the total assemblage.

(4) On the basis of the distribution of live foraminifera, *Ammonia* gr. *tepida* occupies the full salinity range (maximum of 51) recorded in the rivers. *Elphidium* gr. *gunteri*, *E.* gr. *excavatum* and *E.* gr. *advenum* have slightly less salinity tolerance and are only found here at salinities below 44. The dominant porcelaneous species *Quinqueloculina* gr. *seminula*, which in these rivers has a very thin test and small size, has moderate salinity tolerance (24-45).

These findings suggest that few species (e.g. *Ammonia* gr. *tepida*, *Elphidium* gr. *gunteri*, *E.* gr. *excavatum*, *E.* gr. *advenum*) overcome low salinity barriers to habitation in the rivers during winter. These species live permanently in the tidal reaches of the rivers although they probably reproduce only under favorable conditions during summer. Transport or migration within an advancing salt wedge, may explain the distributions of most of the other species found in the rivers with the most diverse assemblages present where a normal-marine salt wedge is maintained in the lower reaches of the river throughout the year (as in the Collie River).

ACKNOWLEDGEMENTS

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APPENDIX 1 LIST OF SPECIES

Species are arranged in alphabetical order. Previous illustrated records from rivers and coastal lagoons in southwest Australia are included below the name used in this study. Because of problems with taxonomy, as revealed for example in the molecular and morphological studies of Hayward *et al.* (2004) for species of *Ammonia*, broad groups are identified for some taxa. Illustrations on Figure 5 include those species that are dominant in the studied rivers, and those on Figure 6 are the less common species that have not been illustrated previously from southwest Australian rivers. Species identifications have been checked against original descriptions and illustrations available in the *Catalogue of Foraminifera* (www.micropress.org).

Ammonia gr. *tepida* (Cushman)
Figure 5 (1–6)
Streblus beccarii (Linné): McKenzie 1962, p. 125, pl. 3, figs. 18, 19: Oyster Harbour.
Ammonia beccarii (Linné): Quilty 1977, p. 90, figs. 45–47: Hardy Inlet.
Ammonia tepida (Cushman): Quilty 1977 p. 90, figs. 48, 49: Hardy Inlet.
Ammonia tepida (Cushman): Revets 2000, p. 371, pl. 3, figs. 16, 17: Leschenault Inlet.
Ammonia aoteana (Finlay): Quilty & Hosie 2006, pl. 1, figs. 21, 22: Swan Estuary.
Ammonia tepida (Cushman): Quilty & Hosie 2006, p. 314, pl. 1, figs. 25, 26: Swan Estuary.

Ammonia gr. *pustulosa* (Albani & Barbero)
Ammonia cf. *pustulosa* (Albani & Barbero): Quilty & Hosie 2006, p. 314, pl. 1, figs. 23, 24: Swan Estuary.

Ammobaculites exiguus Cushman and Bronnimann
Figure 5 (13, 20)
Ammobaculites agglutinans (d'Orbigny): McKenzie 1962, p. 119, pl. 1, fig. 4: Oyster Harbour.
Ammobaculites agglutinans (d'Orbigny): Quilty 1977, p. 89, fig. 10: Hardy Inlet.
Ammobaculites agglutinans (d'Orbigny): Quilty & Hosie 2006, pl. 1, fig. 1: Swan Estuary.

Aubignyna sp.
Figure 6 (1–3)

Bolivina striatula Cushman
Bolivina striatula Cushman: Revets 2000, p. 371, pl. 2, figs. 53, 54: Leschenault Inlet.

Bolivina sp.
Bolivina brittanica Macfadyen: Revets 2000, p. 371, pl, 2, figs. 45, 46: Leschenault Inlet.

Bulimina marginata d'Orbigny
Figure 6 (4)

Buliminella elegantissima d'Orbigny
Figure 6 (5)

Cornuspira planorbis Schultze
Cornuspira planorbis Schultze: Revets 2000, p. 370, pl. 1, figs. 36, 37: Leschenault Inlet.

Elphidium advenum (Cushman)
Figure 5 (9, 10)
Elphidium cf. *advenum* (Cushman) (Figure 5 nos. 11, 12)
Elphidium sp 1: Revets 2000 p. 371, pl 4, figs. 11, 12: Leschenault Inlet.

Elphidium excavatum (Terquem)
Figure 5 (14, 15)

Elphidium cf. *excavatum* (Terquem)
Figure 5 (16, 17)
Elphidium excavatum (Terquem): Quilty & Hosie 2006, p. 314, pl. 1, fig. 28: Swan Estuary.

Elphidium gr. *gunteri* Cole
Figure 5 (7, 8)
Elphidium reticulosum Cushman: Revets 2000, p. 371, pl. 4, figs. 9, 10: Leschenault Inlet.
Elphidium gunteri Cole: Quilty & Hosie 2006, p. 314, pl. 1, fig. 29: Swan Estuary.

Elphidium limbatum (Chapman)
Elphidium limbatum (Chapman): Revets 2000, p. 371, pl. 4, figs. 7, 8: Leschenault Inlet.

Elphidium sp.
This species is represented by very few specimens and is not illustrated.

Fissurina sp.
Fissurina sp.: Revets 2000, p. 371, pl. 2, figs. 40, 41: Leschenault Inlet.

Haplophragmoides sp.
This species is represented by very few specimens and is not illustrated.

Leptohalysis catella (Höglund)
Leptohalysis catella (Höglund): Revets 2000, p. 370, pl. 1, figs. 9, 10: Leschenault Inlet.

Lamellodiscorbis dimidiatus (Jones & Parker)
Discorbis dimidiatus (Jones & Parker): McKenzie 1962, p. 125, pl. 3, figs. 7, 8: Oyster Harbour.
Discorbis dimidiatus (Jones & Parker): Quilty 1977, p. 89, figs. 31, 32: Hardy Inlet.

Miliammina fusca (Brady)
Miliammina fusca (Brady): McKenzie 1962, p. 119, pl. 1, fig. 1: Oyster Harbour.
Miliammina fusca (Brady): Quilty 1977, p. 89, fig. 9: Hardy Inlet.
Miliammina fusca (Brady): Revets 2000, pl. 1, figs 11, 12: Leschenault Inlet.

Neoconorbina sp.
Figure 6 (6, 7)

Nonionellina labradorica (Dawson)
Nonionellina labradorica (Dawson): Revets 2000, p. 371, pl. 4, figs. 33, 34: Leschenault Inlet.

Paratrochammina simplissima (Cushman & McCulloch)
Paratrochammina simplissima (Cushman & McCulloch): Revets 2000, p. 370, pl. 1, figs. 17, 18: Leschenault Inlet.

Paratrochammina sp.
Figure 6 (8–10)

Palustrella earlandi (Parker)
Textularia earlandi Parker: Revets 2000, p. 370, pl. 1 figs. 25, 26: Leschenault Inlet.

Pseudohelenina sp.
This species is represented by very few specimens and is not illustrated.

Quinqueloculina gr. *seminula* (Linnaeus)
Figure 5 (18, 19)
Quinqueloculina seminula (Linnaeus): Revets 2000, p. 370, pl. 2, figs. 3, 4: Leschenault Inlet.
Quinqueloculina seminula (Linné): Quilty 1977, figs 15, 16: Hardy Inlet.

Quinqueloculina littoralis (Collins)
Quinqueloculina striata d'Orbigny: Quilty 1977, p. 89, fig. 20: Hardy Inlet.
Triloculina littoralis Collins: Revets 2000, p. 371, pl. 2, figs. 25, 26: Leschenault Inlet.

Rosalina sp.
Figure 6 (11–13)

Spiroloculina sp.
Figure 6 (14, 15)

Tolypammina sp.
This species is represented by very few specimens and is not illustrated.

Trochammina inflata (Montagu)
Trochammina inflata (Montagu): McKenzie 1962, p. 119, pl. 1, figs. 7, 8: Oyster Harbour.

Trochammina? sp.
Figure 6 (16–18)

Warrenita sp.
Protoshista findens (Parker): Quilty 1977, p. 89, fig. 8: Hardy Inlet.

SUPPLEMENTARY PAPERS

Table 1 Location of samples (latitude and longitude) and water quality measurements: depth (m), temperature (°C), pH, salinity, dissolved oxygen (% saturation).

Table 2 Summary of sediment characteristics.

Table 3 Distribution of living foraminifera, Serpentine River.

Table 4 Distribution of living foraminifera, Murray River.

Table 5 Distribution of living foraminifera, Collie River.

Abundance, distribution and new records of scleractinian corals at Barrow Island and Southern Montebello Islands, Pilbara (Offshore) Bioregion

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The local abundance and distribution of scleractinian corals were documented near Barrow Island in the Pilbara (Offshore) Bioregion, Western Australia. Using a standard rapid ecological assessment method we recorded 204 species from 51 genera and as a result of this study we extend the known distribution range of 15 species. We find a high diversity of habitat types promotes high species richness, particularly among *Acropora* species. Our results confirm the existence of a unique suite of coral species in the Pilbara that is not recorded in the Oceanic Shoals (Offshore) or Pilbara (Nearshore) Bioregions. The Pilbara has a rich coral fauna that is often overlooked and the Barrow/Montebello Islands group may provide a high latitude refuge for some coral species including 39 species that are listed as Vulnerable on the IUCN red list of threatened species.

KEYWORDS: biodiversity, corals, demography, hermatypic, IUCN, local extinction.

INTRODUCTION

The status of coral-reef ecosystems is closely related to the health, abundance and diversity of hermatypic (reef-building) scleractinian corals which engineer the structural framework of the reef, contribute to primary production and nutrient recycling, and provide microhabitat and food for a wide diversity of coral reef species (Done *et al.* 1996; Paulay 1997; Knowlton & Rohwer 2003). In order to effectively safeguard scleractinian (hard) corals, it is important to have baseline demographic data about the structure and composition of communities across multiple scales, because this enables changes to be detected, and helps community responses to impact or management action to be disentangled from natural stochastic processes (Willis & Birks 2006).

In Australia, the majority of coral demographic studies have been undertaken on the east coast (Babcock 1991; Hughes 1996; Connell *et al.* 1997; Hughes *et al.* 1999; DeVantier *et al.* 2006) and the demography of coral communities on the west coast of Australia is less well-documented. Nevertheless, reef systems are well-developed in Western Australia ranging from extensive high-latitude reefs (e.g. Houtman Abrolhos) to immense fringing reef systems (e.g. Ningaloo and the Kimberley) to offshore atolls (e.g. Scott and Seringapatam Reefs). The latitudinal extent of coral communities in Western Australia has expanded and contracted in response to climate change since the Pleistocene. It is predicted that coral species that currently occur in tropical locations will migrate south in response to future climate change and potentially persist in higher latitude 'temperature refugia' (Greenstein & Pandolfi 2008). Thus it is imperative that such potential refuges are identified and protected (Fabricius *et al.* 2007).

For species to migrate south from tropical locations in Western Australia it is necessary for them to pass through the Pilbara region. Currently, coral communities in the Pilbara region are relatively undisturbed due to a low level of urbanisation (DEC 2007). Initial investigations suggest the Pilbara region makes a significant contribution to Western Australia's coral biodiversity (Marsh 1997, 2000; Veron & Marsh 1988; Veron 1993; Blakeway & Radford 2004; Griffith 2004). However, there is a notable lack of quantitative information about the ecological structure of coral communities in this region. Existing studies from inshore locations suggest that the composition of coral communities varies considerably in response to wave exposure, natural turbidity and current movements (Semeniuk *et al.* 1982; Blakeway & Radford 2004). Where wave energy is high and there are low to moderate levels of turbidity, *Acropora* predominates. As the level of turbidity increases, the dominant community shifts towards other types of coral such as *Porites*, *Pavona* and Faviidae, and where turbidity is very high, *Turbinaria* predominates.

Here, we examine the biodiversity and local demographic patterns of scleractinian corals at 14 sites in the vicinity of Barrow Island, located in the Pilbara (Offshore) Bioregion on the North West Shelf of Australia. We report new biogeographic information and highlight the Pilbara as a significant centre of coral diversity at regional, national and international scales. Finally we discuss demographic patterns that are relevant to the effective management of coral biodiversity.

METHODS

Site description

Barrow Island lies ~1200 km north of Perth and 56 km from the coastline on the North West Shelf of Western

Australia (20.82°S, 115.39°E). Barrow Island is the largest of a group of islands (which includes the Montebello and Lowendal Islands) that are formed by a long shallow bank of Miocene limestone with Pleistocene carbonate reef deposits (Veron & Marsh 1988).

The Barrow, Montebello and Lowendal Islands occur within the Pilbara (Offshore) (PIO) Bioregion (IMCRA 2006). The PIO Bioregion occurs in waters seaward of 10 m depth contour between North West Cape and the Montebello Islands (Figure 1). The PIO Bioregion occurs offshore from the Pilbara (Nearshore) Bioregion, although it is not to be confused with the Oceanic Shoals (Offshore) Bioregion which occurs seaward of the North West Shelf.

Coral-reef ecosystems in the PIO Bioregion are considered unique due to the complexity of substrate types, oceanographic conditions and habitat diversity (IMCRA 1998; Brewer *et al.* 2007; DEC 2007). Tides in the Barrow Island region are semidiurnal, (two high tides and two low tides per day) and the tidal range varies significantly around the island with a maximum spring tide on the east coast over 4 m and <2.5 m on the west coast. Sea-surface temperatures in the area vary from 23°C to 29°C (Chevron Australia 2011) and there are frequent cyclones in the region (e.g. four tropical cyclones were recorded near Barrow Island during the 2010/2011 cyclone season).

Survey design and analysis

Coral-biodiversity surveys were conducted at 14 sites in the vicinity of Barrow Island. Twelve sites were surveyed along a north–south gradient on the eastern side of Barrow Island, one protected back-reef habitat was surveyed on the western side of the island and one site was surveyed at the southern Montebello Islands (Figure 2; Appendix 1). A standard method of rapid ecological assessment (DeVantier *et al.* 1998; Kospartov *et al.* 2006) was undertaken by snorkel. Coral communities were documented to a maximum depth of 10 m, encompassing an area of ~150 m radius. At each site, surveys were conducted for 120 minutes, or until species saturation was reached (i.e. until no new species were recorded for a 15 minute period).

Species were identified *in situ* to species level wherever possible; otherwise a voucher specimen was collected for subsequent identification at the Museum of Tropical Queensland. Species names were interpreted according to Veron (2000) and Wallace (1999). One unidentified species has been designated as *Acropora arafura*, a new species recently described by Carden Wallace (Wallace *et al.* 2012).

The relative abundance of each species was ranked on a five-point scale adapted from a DAFOR scale which is commonly used in flora and fauna surveys and broadly analogous to a logarithmic scale: 1, rare (1, 2 colonies); 2, infrequent (3–5 colonies); 3, frequent (6–20 colonies); 4, common (21–50 colonies); 5, dominant (51+ colonies) (Jongman *et al.* 1995; DeVantier *et al.* 1998). These ordered variables have a relative magnitude interpretation, so a mean difference of 1 corresponds approximately to a log difference in abundance. When large stands were encountered, every 1 m² was treated as two colonies.

Figure 1 Map of the bioregions in north Western Australia as delineated by IMCRA (1998). Bioregions are: OSS, Oceanic Shoals; KIM, Kimberley; NWS, North West Shelf; KSD, King Sound; CAN, Canning; EMB, Eighty Mile Beach; PIN, Pilbara (Nearshore); PIO, Pilbara (Offshore); NIN, Ningaloo; ZUY, Zuytdorp; SBY, Shark Bay.
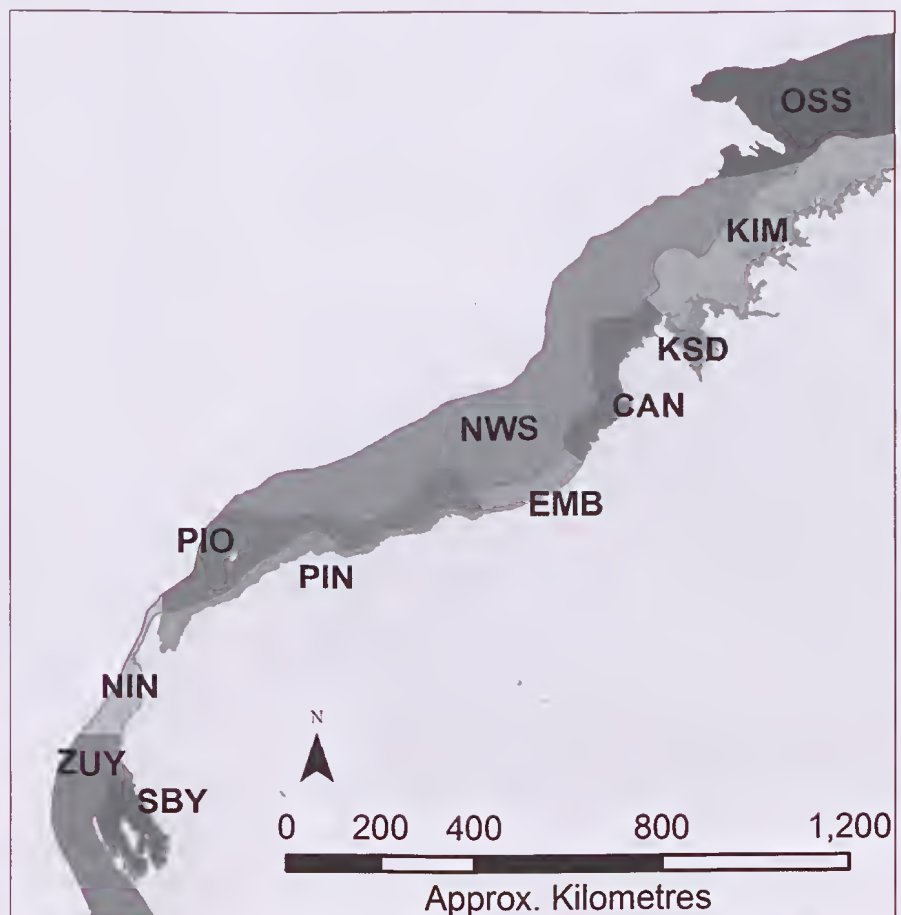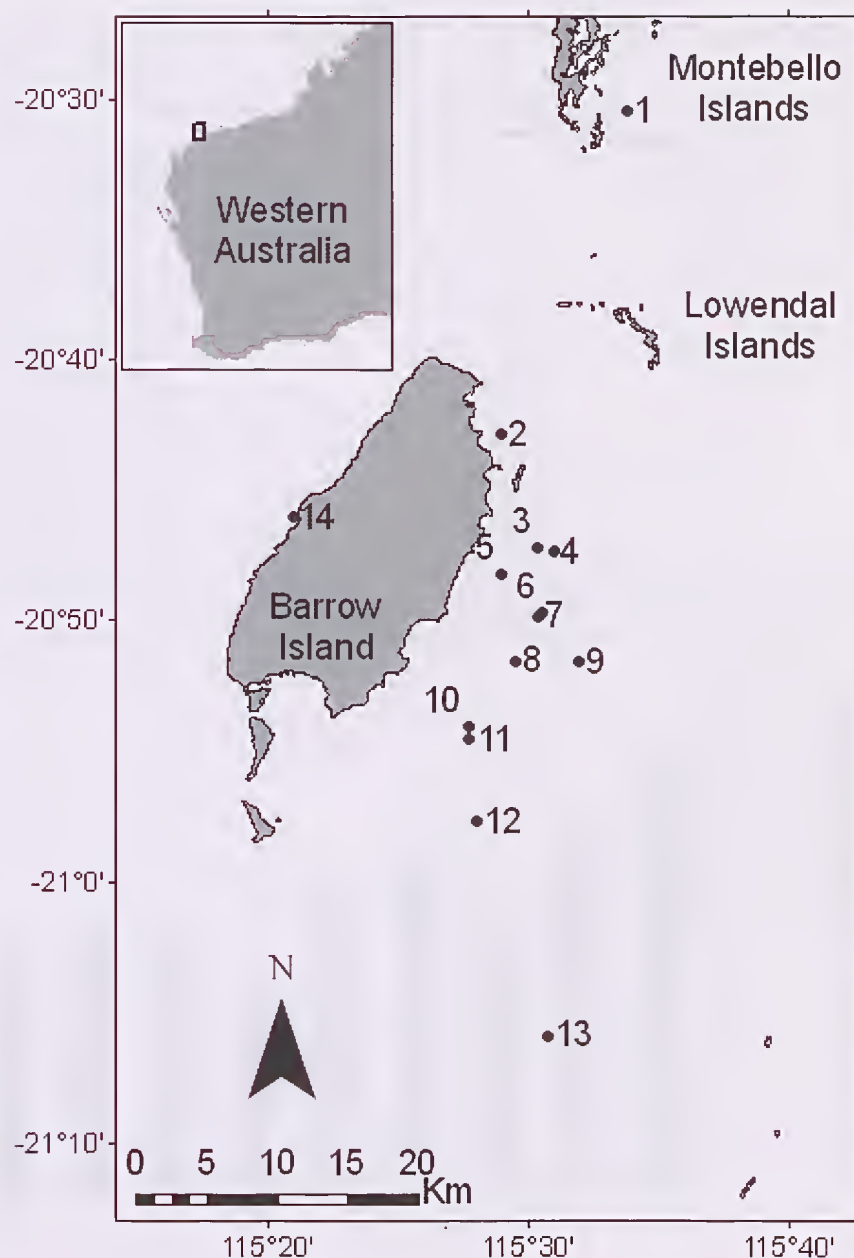
Figure 2 Map of study sites in the vicinity of Barrow/Montebello Islands that are located 56 km off the Pilbara coast on the North West Shelf of Western Australia.

Species accumulation curves were calculated for each location using the 'vegan' library in R using the function 'specaccum' with jack-knifed standard errors (Oksanen *et al.* 2009). This provided a graphical check of whether sampling was sufficient to detect rare members of the assemblage. The threatened status of coral species were downloaded from the IUCN Red List of threatened species (www.iucnredlist.org).

RESULTS

A total of 204 species from 51 genera of scleractinian corals was recorded (Tables 1, 2). We report here 15 new taxonomic records (Table 3), including five for Australia, five for Western Australia and five for the North West Shelf. The species accumulation curve approached an asymptote indicating this dataset provides a good representation of the coral diversity in shallow reef locations near Barrow Island (Figure 3), although additional survey sites are likely to be needed to fully represent the wider diversity present in the P1O Bioregion because an additional 41 species recorded by Marsh (2000) from the Montebello Islands were not recorded in the present survey.

Site-specific species richness varied from a minimum of 50 species along the western side of Barrow Island to a maximum of 103 species at the southern Montebello Islands (Figure 4). The level of diversity along the eastern side of Barrow Island was relatively homogeneous but the proportional composition of different genera changed from *Acropora*-dominated communities in the north to *Porites*-dominated communities in the south.

Except for Site 14 on the west coast, characteristic fringing reef habitats such as reef flats and slopes were not immediately evident. Moreover survey sites consisted of isolated patch reef habitat with low-medium rugosity surrounded by soft-bottom inter-reefal habitat dominated by *Echinopora, Merulina* and *Pectinia*. Two sites (2 and 3)
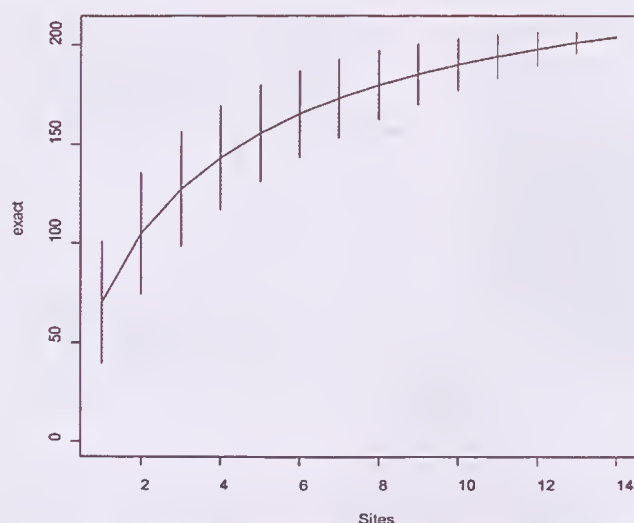
Figure 3 Species accumulation curve shows that this dataset is a good representation of the coral diversity in the vicinity of Barrow/Montebello Islands as the species accumulation curve approaches an asymptote.
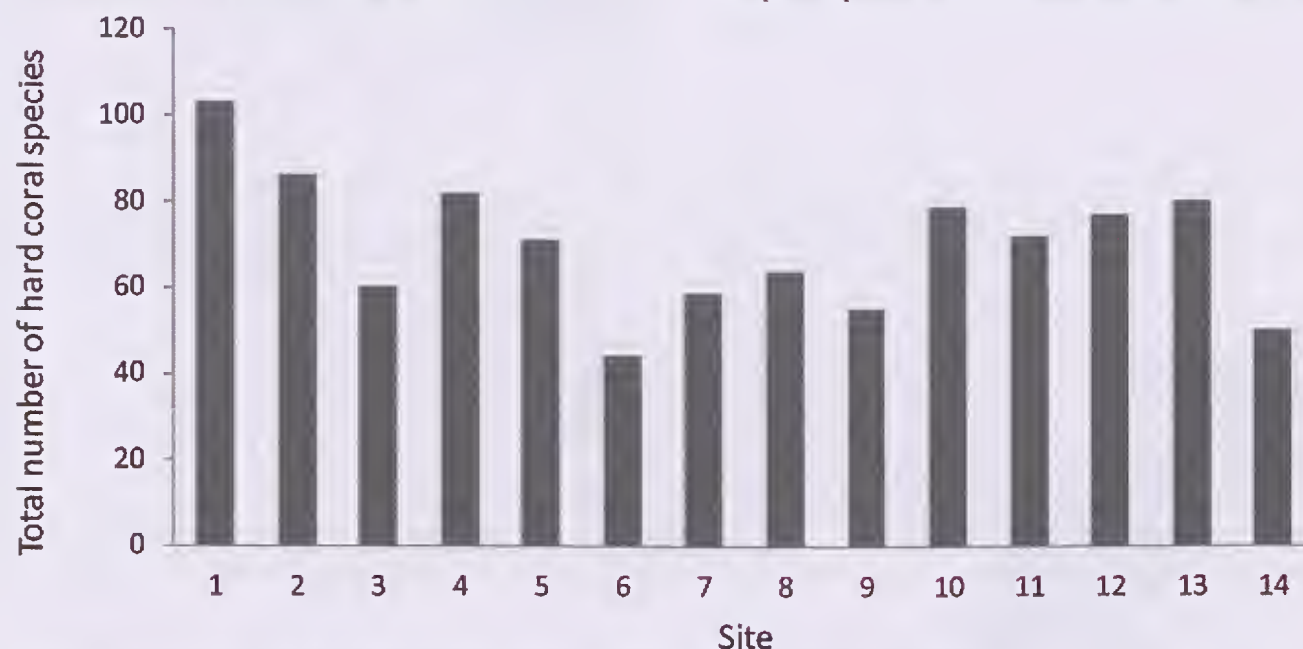
in the northeast of the island were dominated by corymbose *Acropora* and branching *Acropora* thickets. Of particular interest are sites on the southeast of the island where very large *Porites* colonies (>8 m greater diameter) were common. At these *Porites*-dominated sites, additional species (such as faviids, mussids and corymbose acroporids) colonised bare parts of the upper colony surfaces.

Of the 51 genera recorded, the most species-rich genera were *Acropora* (47 spp.), *Montipora* (19 spp.) and *Favia* (12 spp.) (Figure 5). The most abundant genera (in terms of the number of colonies) were *Acropora*, *Porites* and *Montipora*. Two species were recorded at all 14 sites (*Lobophyllia hemprichii* and *Pocillopora damicornis*); however they did not predominate at any site (Table 2). Species that were numerically dominant (i.e. over 51 colonies per site) and locally widespread (recorded at 10 or more sites) were *Porites lutea, Echinopora lamellosa, Merulina ampliata* and *Pectinia lactuca*. Four other species—*Porites australiensis, Acropora muricata, Acropora nasuta* and *Porites cylindrica*—were also locally widespread (present at 10 or more sites) and sometimes

Figure 4 Species richness at the 14 survey sites in the vicinity of Barrow/Montebello Islands.

Table 1 Total number of hard coral species recorded in different regions of Western Australia.

Location	Lat/long	Species richness	Source
Ashmore Reef	12°15'S 123°02'E	275	Richards *et al.* 2009
Scott Reef	14°10'S 121°84'E	297	McKinney 2006
Bonaparte Archipelago	14°17'S 125°18'E	278	Z Richards, C Wallace, P Muir, J Veron, A Noreen, N Rosser & M Forde unpubl. data.
Rowley Shoals	17°20'S 119°20'E	188	Veron 2004
Dampier Archipelago	20°32'S 116°36'E	229	Griffith 2004
Montebello Islands	20°28'S 115°33'E	150	Marsh 2000
Barrow Island/South Montebello Island	20°82'S 115°39'E	204	This survey
Ningaloo Reef	22°40'S 113°39'E	217	Veron & Marsh 1988
Shark Bay	25°30'S 113°30'E	81	Veron 2004
Abrolhos Islands	28°43'S 113°47'E	192	Veron 2004
Geraldton	28°46'S 114°36'E	14	Veron 2004
Rottnest Island	32°00'S 115°31'E	19	Veron 2004

Table 2 The corals of Barrow Island/South Montebello Island with local pattern of occupancy denoted according to the number of sites occupied and the relative abundance at each site.

Species	No. of sites	Relative abundance	IUCN classification
Family Acroporidae			
Acropora aculeus (Dana 1846)	1	rare	Vu
Acropora acuminata (Verrill 1864)	3	rare to infrequent	Vu
Acropora anthocercis (Brook 1893)	1	rare	Vu
Acropora austera (Dana 1846)	8	rare to dominant	NT
Acropora cerealis (Dana 1846)	3	rare to infrequent	LC
Acropora arafura (Wallace, Done & Muir 2012)	9	rare to infrequent	Not assessed
Acropora cytherea (Dana 1846)	1	rare	LC
Acropora digitifera (Dana 1846)	3	rare to frequent	NT
Acropora divaricata (Dana 1846)	13	rare to frequent	NT
Acropora donei Veron& Wallace 1984	2	rare	Vu
Acropora florida (Dana 1846)	11	rare to frequent	NT
Acropora gemmifera (Brook 1892)	4	rare to infrequent	LC
Acropora glauca (Brook 1893)	4	rare to infrequent	NT
Acropora graudis (Brook 1892)	4	rare to infrequent	LC
Acropora horrida (Dana 1846)	1	rare	Vu
Acropora humilis (Dana 1846)	3	rare	NT
Acropora hyacinthus (Dana 1846)	5	rare to frequent	NT
Acropora insignis Nemenzo 1967	5	rare	DD
Acropora intermedia (Brook 1891)	6	rare to dominant	LC
Acropora latistella (Brook 1891)	4	rare	LC
Acropora listeri (Brook 1893)	1	rare	Vu
Acropora loripes (Brook 1892)	3	rare to infrequent	NT
Acropora lovelli Veron & Wallace 1984	3	rare	Vu
Acropora lutkeni Crossland 1952	4	rare	NT
Acropora microclados (Ehrenberg 1834)	1	rare	Vu
Acropora microphthalma (Verrill 1859)	2	rare to infrequent	LC
Acropora millepora (Ehrenberg 1834)	8	rare to common	NT
Acropora muricata (Linnaeus 1758)	11	rare to common	NT
Acropora nasuta (Dana 1846)	11	rare to common	NT
Acropora palmerae Wells 1954	1	rare	Vu
Acropora polystoma (Brook 1891)	2	rare	Vu
Acropora pulchra (Brook 1891)	3	infrequent	LC
Acropora robusta (Dana 1846)	2	rare	LC
Acropora samoensis (Brook 1891)	10	rare to infrequent	LC
Acropora sarmentosa (Brook 1892)	1	rare	LC
Acropora secale (Studer 1878)	3	rare to infrequent	NT
Acropora selago (Studer 1878)	3	rare	NT
Acropora solitaryensis Veron & Wallace 1984	3	rare	Vu
Acropora spicifera (Dana 1846)	7	rare to frequent	Vu
Acropora subulata (Dana 1846)	3	rare	LC
Acropora tenuis (Dana 1846)	10	rare to frequent	NT
Acropora valenciennesi (Milne Edwards & Haime 1860)	6	rare to infrequent	LC
Acropora valida (Dana 1846)	3	rare to infrequent	LC
Acropora verweyi Veron & Wallace 1984	1	rare	Vu
Acropora willisae Veron & Wallace 1984	3	rare to infrequent	Vu
Acropora yongei Veron & Wallace 1984	1	rare	LC
Astreopora gracilis Bernard 1896	1	infrequent	LC
Astreopora listeri Bernard 1896	2	rare	LC
Astreopora myriophthalma (Lamarck 1816)	6	rare to frequent	LC
Isopora brueggemanni (Brook 1893)	4	rare to frequent	Vu
Montipora aequituberculata Bernard 1897	7	rare to frequent	LC
Montipora crassituberculata Bernard 1897	3	rare to infrequent	Vu
Montipora danae (Milne Edwards & Haime 1851)	4	rare	LC
Montipora digitata (Dana 1846)	3	rare to frequent	LC
Montipora efflorescens Bernard 1897	4	rare	NT
Montipora foliosa (Pallas 1766)	1	rare	NT
Montipora grisea Bernard 1897	2	rare to infrequent	LC
Montipora hispida (Dana 1846)	8	rare to frequent	LC
Montipora incrassata (Dana 1846)	1	rare	NT
Montipora informis Bernard 1897	1	rare	LC
Montipora mollis Bernard 1897	2	rare	LC
Montipora monasteriata (Forskål 1775)	2	rare	LC

Table 2 (cont.)

Species	No. of sites	Relative abundance	IUCN classification
Montipora peltiformis Bernard 1897	6	rare to infrequent	NT
Montipora stellata Bernard 1897	1	rare	LC
Montipora tuberculosa (Lamarck 1816)	6	rare to infrequent	LC
Montipora turgescens Bernard 1897	5	rare to infrequent	LC
Montipora turtlensis Veron & Wallace 1984	2	rare	Vu
Montipora undata Bernard 1897	6	rare to infrequent	NT
Montipora verrucosa (Lamarck 1816)	3	rare	LC
Family Agariciidae			
Gardineroseris planulata Dana 1846	1	frequent	LC
Leptoseris explanata Yabe & Sugiyama 1941	1	rare	LC
Leptoseris mycetoseroides Wells 1954	1	rare	LC
Pachyseris rugosa (Lamarck 1801)	6	rare to infrequent	Vu
Pachyseris speciosa (Dana 1846)	9	infrequent to common	LC
Pavona clavus (Dana 1846)	2	rare	LC
Pavona decussata (Dana 1846)	5	rare to infrequent	Vu
Pavona duerdeni Vaughan 1907	2	rare	LC
Pavona explanulata (Lamarck 1816)	2	rare to infrequent	LC
Pavona maldivensis (Gardiner 1905)	1	rare	LC
Pavona varians Verrill 1864	2	rare to infrequent	LC
Pavona venosa (Ehrenberg 1834)	1	rare	Vu
Family Dendrophyliidae			
Turbinaria bifrons Brüggemann 1877	2	rare	Vu
Turbinaria mesenterina (Lamarck 1816)	6	rare to frequent	Vu
Turbinaria patula (Dana 1846)	2	rare	Vu
Turbinaria peltata (Esper 1794)	2	rare	Vu
Turbinaria reniformis Bernard 1896	10	rare to frequent	Vu
Family Euphylliidae			
Euphyllia ancora Veron & Pichon 1979	2	rare	Vu
Euphyllia glabrescens (Chamisso & Eysenhardt 1821)	1	rare	NT
Physogyra lichtensteini (Milne Edwards & Haime 1851)	2	rare	Vu
Plerogyra sinuosa (Dana, 1846)	3	rare	NT
Family Faviidae			
Caulastrea curvata Wijsmann-Best 1972	5	rare to frequent	Vu
Cyphastrea chalcidium (Forskål 1775)	5	rare to infrequent	LC
Cyphastrea microphthalma (Lamarck 1816)	12	rare to frequent	LC
Cyphastrea serailia (Forskål 1775)	3	rare to infrequent	LC
Diploastrea heliopora (Lamarck 1816)	5	rare to common	NT
Echinopora ashmorensis Veron 1990	4	rare to frequent	Vu
Echinopora lamellosa (Esper 1795)	12	rare to dominant	LC
Favia favus (Forskål 1775)	3	rare to infrequent	LC
Favia helianthoides Wells 1954	3	rare	NT
Favia laxa (Klunzinger 1879)	2	rare	NT
Favia maritima (Nemenzo 1971)	2	rare	NT
Favia matthaii Vaughan 1918	4	rare to infrequent	NT
Favia maxima Veron, Pichon & Wijsman-Best 1972	1	rare	NT
Favia pallida (Dana 1846)	11	rare to frequent	LC
Favia rotumana (Gardiner 1899)	4	rare	LC
Favia rotundata Veron, Pichon & Wijsman-Best 1972	8	rare	NT
Favia speciosa Dana 1846	8	rare to frequent	LC
Favia stelligera (Dana 1846)	4	rare to infrequent	NT
Favia veroni Moll & Borel-Best 1984	4	rare	NT
Favites abdita (Ellis & Solander 1786)	10	rare to infrequent	NT
Favites acuticollis (Ortmann 1889)	1	infrequent	NT
Favites chinensis (Verrill 1866)	3	rare	NT
Favites complanata (Ehrenberg 1834)	4	rare	NT
Favites flexuosa (Dana 1846)	2	rare	NT
Favites halicora (Ehrenberg 1834)	12	rare to frequent	NT
Favites paraflexuosa Veron 2000	1	rare	NT
Favites pentagona (Esper 1794)	5	rare to infrequent	LC
Favites russelli (Wells 1954)	3	rare	NT
Favites stylifera (Yabe & Sugiyama 1937)	3	rare	NT
Goniastreaaspera Verrill 1905	3	rare to frequent	LC
Goniastrea australensis (Milne Edwards & Haime 1857)	9	rare to infrequent	LC

Species	No. of sites	Relative abundance	IUCN classification
Goniastrea edwardsi Chevalier 1971	3	rare to infrequent	LC
Goniastrea favulus (Dana 1846)	3	rare to frequent	NT
Goniastrea pectinata (Ehrenberg 1834)	12	rare to frequent	LC
Goniastrea retiformis (Lamarck 1816)	9	rare to common	LC
Leptastrea pruinosa Crossland 1952	1	rare	LC
Leptastrea purpurea (Dana 1846)	5	rare	LC
Leptastrea transversa Klunzinger 1879	2	rare	LC
Leptoria phrygia (Ellis & Solander 1786)	5	rare to infrequent	NT
Montastrea colemani Veron 2000	2	rare	NT
Montastrea curta (Dana 1846)	9	rare to infrequent	LC
Montastrea salebrosa (Nemenzo 1959)	1	rare	Vu
Moseleya latistellata Quelch 1884	2	rare	Vu
Oulophyllia bennettae (Veron & Pichon 1977)	3	rare	NT
Oulophyllia crispa (Lamarck 1816)	3	rare	NT
Platygyra acuta Veron 2000	9	rare	NT
Platygyra daedalea (Ellis & Solander 1786)	10	rare to frequent	LC
Platygyra lamellina (Ehrenberg 1834)	8	rare to infrequent	NT
Platygyra pini Chevalier 1975	13	rare to frequent	LC
Platygyra ryukyuensis Yabe & Sugiyama 1936	5	rare to infrequent	NT
Platygyra sinensis (Milne Edwards & Haime 1849)	13	rare to infrequent	LC
Platygyra yaeyamaensis Eguchi & Shirai 1977	1	rare	Vu
Plesiastrea versipora (Lamarck 1816)	3	rare	LC
Family Fungiidae			
Fungia scruposa Klunzinger 1879	1	infrequent	LC
Fungia fungites (Linneaus 1758)	1	frequent	NT
Fungia repanda Dana 1846	9	rare to frequent	LC
Halomitra pileus (Linnaeus 1758)	1	rare	LC
Herpolitha limax (Houttuyn 1772)	8	rare to infrequent	LC
Lithophyllon undulatum Rehberg 1892	9	rare to infrequent	NT
Podabacia crustacea (Pallas 1766)	8	rare to frequent	LC
Polyphyllia talpina (Lamarck 1801)	1	rare	
Family Merulinidae			
Hydnophora exesa (Pallas 1766)	11	rare to infrequent	NT
Hydnophora grandis Gardiner 1904	6	rare to infrequent	LC
Hydnophora microconos (Lamarck 1816)	1	rare	NT
Hydnophora pilosa Veron 1985	9	rare to frequent	LC
Hydnophora rigida (Dana 1846)	5	rare to infrequent	LC
Merulina ampliata (Ellis & Solander 1786)	12	rare to dominant	LC
Merulina scabricula Dana 1846	7	rare to frequent	LC
Scapophyllia cylindrica Milne Edwards & Haime 1848	1	rare	LC
Family Mussidae			
Acanthastrea echinata (Dana 1846)	7	rare	LC
Acanthastrea hemprichii (Ehrenberg 1834)	2	rare	Vu
Acanthastrea subechinata Veron 2000	1	rare	NT
Blastomussa merleti Wells 1961	2	rare	LC
Lobophyllia corymbosa (Forskål 1775)	12	rare to infrequent	LC
Lobophyllia diminuta Veron 1985	6	rare to frequent	Vu
Lobophyllia flabelliformis Veron 2000	2	rare	Vu
Lobophyllia hemprichii (Ehrenberg 1834)	14	rare to frequent	LC
Lobophyllia robusta Yabe & Sugiyama 1936	5	rare to infrequent	LC
Scolymia australis (Milne Edwards & Haime 1849)	1	rare	LC
Symphyllia radians Milne Edwards & Haime 1849	3	rare	LC
Symphyllia recta (Dana 1846)	5	rare	LC
Family Oculinidae			
Galaxea astreata (Lamarck 1816)	13	rare to frequent	Vu
Galaxea fascicularis (Linnaeus 1767)	10	rare to frequent	NT
Family Pectiniidae			
Echinophyllia aspera (Ellis & Solander 1788)	9	rare to infrequent	LC
Echinophyllia orpheensis Veron & Pichon 1980	5	rare to infrequent	LC
Mycedium elephantotus (Pallas 1766)	5	rare to infrequent	LC
Mycedium robokaki Moll & Borel-Best 1984	1	rare	LC
Oxypora glabra Nemenzo 1959	7	rare to frequent	LC
Oxypora lacera Verrill 1864	11	rare to infrequent	LC
Pectinia lactuca (Pallas 1766)	12	rare to dominant	Vu
Pectinia paeonia (Dana 1846)	6	rare to common	NT

Table 2 (cont.)

Species	No. of sites	Relative abundance	IUCN classification
Family Pocilloporidae			
Pocillopora damicornis (Linnaeus 1758)	14	rare to frequent	LC
Pocillopora verrucosa (Ellis & Solander 1786)	5	rare to infrequent	LC
Seriatopora caliendrum Ehrenberg 1834	7	rare to frequent	NT
Stylophora pistillata Esper 1797	6	rare to frequent	NT
Stylophora subseriata (Ehrenberg 1834)	1	rare	LC
Family Poritidae			
Goniopora burgosi Nemenzo 1955	1	rare	Vu
Goniopora lobata Milne Edwards & Haime 1860	8	rare to infrequent	NT
Goniopora stokesi Milne Edwards & Haime 1851	2	rare	NT
Goniopora tenuidens (Quelch 1886)	2	rare	LC
Porites annae Crossland 1952	10	rare to infrequent	NT
Porites australiensis Vaughan 1918	12	rare to common	LC
Porites cylindrica Dana 1846	10	infrequent to common	NT
Porites lichen Dana 1846	5	rare to frequent	LC
Porites lutea Milne Edwards & Haime 1851	13	infrequent to dominant	LC
Porites nigrescens Dana 1846	9	rare to frequent	Vu
Porites rus (Forskål 1775)	5	rare to frequent	LC
Porites solida (Forskål 1775)	2	rare to infrequent	LC
Family Siderasteridae			
Coscinaraea columna (Dana 1846)	4	rare	LC
Psammocora contigua (Esper 1797)	10	rare to frequent	NT
Psammocora digitata Milne Edwards & Haime 1851	5	rare	NT
Psammocora nierstraszi van der host 1921	1	rare	LC
Psammocora obtusangula (Lamarck 1816)	1	rare	NT

[a] The global threatened status of each species is noted according to IUCN categories and criteria: Vu, vulnerable; NT, near threatened; LC, least concern; DD, data deficient.

Table 3 New taxonomic records identified during these surveys.

Species	New record	Sites where species recorded
Acanthastrea subechinata Veron 2000	Australia	12
Favites acuticollis (Ortmann 1889)	Australia	14
Platygyra acuta Veron 2000	Australia	1,5,6,7,10,11,12,13,14
Platygyra yaeyamaensis Eguchi & Shirai 1977	Australia	1
Stylophora subseriata (Ehrenberg 1834)	Australia	1
Acanthastrea hemprichii (Ehrenberg 1834)	Western Australia	2,5
Hydnophora grandis Gardiner 1904	Western Australia	1,4,8,9,10,11
Lobophyllia robusta Yabe & Sugiyama 1936	Western Australia	4,5,10,11,13
Montastrea colemani Veron 2000	Western Australia	1,8
Montastrea salebrosa (Nemenzo 1959)	Western Australia	1
Psammocora obtusangula (Lamarck 1816)	North West Shelf	1
Goniopora burgosi Nemenzo 1955	North West Shelf	12
Mycedium robokaki Moll & Borel-Best 1984	North West Shelf	4
Pavona duerdeni Vaughan 1907	North West Shelf	10,11
Scolymia australis (Milne Edwards & Haime 1849)	North West Shelf	1

Table 4 Percentage of species in each abundance category

Abundance category	Percentage of species
Rare	44.1
Rare to infrequent	25.7
Rare to frequent	19.8
Rare to common	3.5
Rare to dominant	2.5
Infrequent	2.0
Infrequent to common	1.0
Frequent	1.0
Infrequent to dominant	0.5

common (between 21 and 50 colonies per site). Fifty-three percent of species were recorded as rare at the sites they occupied, while 32% of species were infrequent, 12% were frequent, 1.5% were common and only 1.2% were dominant (Table 4). Eighty-nine species were always recorded as rare and 35 of these rare species (17%) were recorded at a single site only (Table 2).

Coral genera that so far have not been recorded from the Barrow/Montebello Islands but are found on other reefs in Western Australia include *Anacropora, Stylocoeniella, Oulastrea, Cantharellus, Diaseris, Heliofungia, Sandolithia* and *Palauastrea*. However, it is important to

Figure 5 Number of species within each genus recorded in the present survey clearly shows *Acropora* and *Montipora* are the most species-rich genera followed by *Favia*, *Favites*, *Porites*, *Pavona* and *Platygyra*.

note that rare or cryptic species or those occurring beyond 10 m depth may not have been detected by our survey methodology. Thus, further surveys, particularly of the reef edge and reef slope habitats on the west coast of Barrow Island, may reveal additional species.

DISCUSSION

This study is the first to record the relative abundance of corals in the PIO Bioregion in Western Australia and describes a community that, when compared with others in the region, has greater species diversity than currently recorded from the Pilbara (nearshore) Bioregion, but lower diversity than that recorded from the offshore atolls and Kimberley (Table 1). Importantly, this community contains 15 species that are not currently represented elsewhere in Western Australia, and in some cases, Australia (Table 3). A total of 204 species from 51 genera of scleractinian coral were recorded in this survey, and following Marsh (1997, 2000) raised the number of scleractinian species recorded from Barrow/Montebello Islands to 245. Thus, the coral communities of the PIO Bioregion make a significant contribution to Australia's biodiversity.

An important finding arising from the provision of abundance data in this study is that of the 204 species recorded in this survey, only eight species (4%) dominate the community (i.e. they were present at 10 or more sites with over 21 colonies counted at one or more sites: *Porites lutea*, *Echinopora lamellosa*, *Merulina ampliata*, *Pectinia lactuca*, *Porites australiensis*, *Acropora muricata*, *Acropora nasuta* and *Porites cylindrica*). Unfortunately few quantitative datasets exist to describe local patterns of species abundance in coral communities at other Pilbara locations, hence it is not currently possible to equivocally

comment about whether this particular suite of dominant species is unique to this location, or whether these species reach a similar abundance elsewhere. Nevertheless, it is likely that the Barrow/Montebello Islands are important centres of abundance for these species.

In a global context, one of these dominant species (*Pectinia lactuca*) is classified as Vulnerable to extinction this century according to IUCN categories and criteria (Carpenter *et al.* 2008), hence the Barrow/Montebello Islands may prove to be a globally important sanctuary for this species. Thirty-eight other members of the Barrow/Montebello Island assemblage are listed as Vulnerable by the IUCN (Table 2) including four of the species whose range has been extended to the PIO Bioregion (*Acanthastrea hemprichii*; *Platygyra yaeyamansis*; *Montastrea salebrosa* and *Goniopora burgosi*). Other species of interest in the Barrow/Montebello Island community that are classified as vulnerable by the IUCN are *Acropora spicifera* and *Echinopora ashmorensis* which, in Australia, are known only from the west coast. All five of the *Turbinaria* spp. documented at Barrow/Montebello Islands are classified as Vulnerable, which is somewhat surprising as *Turbinaria* are often described as a hardy species (Fabricius *et al.* 2007).

Our results suggest that almost half of the coral species in the vicinity of Barrow Island have very small population sizes. Eighty-nine species (44%) recorded in this survey were rare (i.e. only one or two colonies recorded per site) (Table 4) and 35 species (17%) were not only rare, but occurred at a single site only (Table 2). Considering the majority of species in ecological communities are rare (Magurran & Henderson 2003), and that some populations of rare species persist over evolutionary time in low numbers, at least partly because their small local populations minimise density-

dependant processes (e.g. disease outbreaks often occur among common species: Gaston 1994), it is not unusual to find a large number of rare species.

The suite of species recorded in this study suggests that there are marked cross-shelf differences in the composition of Pilbara communities. For example, at inshore sites near the Passage Island group (Onslow coastline), *Turbinaria, Favites, Platygyra, Goniopora* and *Lobophyllia* were the most abundant genera and there is very little representation of *Acropora* and *Pavona* (Z T Richards unpubl. data). At Barrow/Montebello Islands, the most abundant genera were *Acropora, Montipora* and *Porites*. These differences in community composition are most likely due to wave exposure and the level of suspended particulate matter in the water column (Marsh 1997; Blakeway & Radford 2005). While terrestrial run-off to the marine environment is generally low in the Pilbara region, input from rivers, and re-suspension of deposited sediments produces extremely turbid conditions in some areas (Margvelashvili *et al.* 2006). Given the dynamic and turbid conditions, it is apparent the many of the corals present in the Pilbara have adapted to a high sediment regime. Even though corals were traditionally thought to prefer clear water because light penetration benefits photosynthesis for their symbiotic dinoflagellates, new studies suggest that at high particle loads, corals gain energy by increasing their heterotrophic feeding (Anthony & Fabricius 2000). Hence, corals growing in the turbid areas of the Pilbara may offset the stress that accompanies high turbidity by changing their trophic mode, which sustains a positive energy balance in highly turbid conditions.

This survey adds many new species records to Western Australia, and highlights the fact that the Pilbara region has a rich coral fauna that is often overlooked. The diversity of habitats and isolated nature of many of the reefs around Barrow Island may provide a refuge for a number of coral species; however considering this community is dominated by only a small number of species with the majority of species occurring in low abundance, conservation plans for this region should be underpinned with quantitative data to accurately predict and protect species at risk.

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APPENDIX 1 GPS CO-ORDINATES AND DEPTH OF EACH OF THE 14 STUDY SITES.

Site	Latitude (S)	Longitude (E)	Approximate water depth (m)
1	20°30.472'	115°33.829'	6.5
2	20°42.898'	115°29.001'	4.0
3	20°47.229'	115°30.363'	3.0
4	20°47.350'	115°30.477'	5.0
5	20°48.249'	115°28.961'	6.0
6	20°49.713'	115°30.507'	9.0
7	20°49.867'	115°30.384'	8.8
8	20°51.575'	115°29.544'	6.5
9	20°51.624'	115°31.976'	9.3
10	20°54.085'	115°27.755'	6.3
11	20°54.949'	115°27.756'	2.5
12	20°57.717'	115°28.067'	3.5
13	21°5.929'	115°30.810'	4.8
14	20°46.068'	115°21.001'	1.5

Searching for signs of bilby (*Macrotis lagotis*) activity in central Western Australia using observers on horseback

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We tested the effectiveness of observers on horseback for searching for signs of activity of bilbies reintroduced to a former pastoral lease in central Western Australia. As a means of transport, horses were able to traverse country not readily accessible to motorised transport including all-terrain vehicles, were productive in terms of area searched per unit time, were an elevated platform from which to observe signs of activity, and had a low impact on soil and vegetation. In this trial, a distance of 27 km was traversed with observers visually inspecting 121.5 ha for signs of bilby activity. Ten fresh (active) bilby burrows were detected, equating to an active burrow every 12.15 ha. Further research will focus on estimating bilby population from the number of active burrows.

KEYWORDS: arid zone, bilby, fauna conservation, *Macrotis lagotis*, translocations.

INTRODUCTION

The greater bilby (*Macrotis lagotis*) is an omnivorous, burrowing marsupial that was distributed throughout ~70% of Australia prior to European settlement. Its population has since significantly declined and its distribution has contracted into the driest and least fertile parts of its former range (Watts 1969; Southgate 1990). Over the period 2007–2010, 128 captive-bred bilbies were reintroduced to Lorna Glen, a former pastoral lease in central Western Australia some 160 km northeast of the town of Wiluna (Pertuisel 2010). The reintroductions were preceded by four years of introduced predator control and other conservation activities associated with Rangelands Restoration, a project jointly managed by the Department of Environment and Conservation (DEC) and the Wiluna Aboriginal Community (Miller *et al.* 2010).

Success of fauna reintroductions is determined by the long-term persistence and breeding of the reintroduced species. It is not feasible to obtain counts of individual bilbies in these landscapes (Southgate *et al.* 2005), but measures of activity can give estimates of relative abundance and microhabitat preferences. In the first few months after release, a group of bilbies was closely monitored by radio tagging and associated cage trapping to obtain information about survivorship of founder animals. Bilbies are no longer being radio-tagged and have dispersed from their points of release. As reported by Southgate *et al.* (1995), bilbies are difficult to monitor by trapping or spotlighting because of their low density, mostly solitary behaviour and great mobility. When bilbies were initially reintroduced at Lorna Glen, localised trapping in areas where they were known to occur was successful, but as the animals dispersed, trapping was not an effective way of assessing bilby

populations due to extremely low trap success (Pertuisel 2010). A feasible alternative was to use tracks, diggings, scats and burrows as a surrogate for their occurrence and relative abundance.

Prior to this trial, we searched for signs of bilby activity using all-terrain vehicles (ATVs) on and off tracks, and observers on foot. Neither technique was found to be practical; ATVs were impeded by thick scrub, fallen timber beneath mulga groves, dense spinifex and recently burnt spinifex. They caused unacceptable crushing damage to vegetation and sensitive soil cryptogamic crusts, and were restricted by vehicle maintenance issues (punctures, grass seed blockages, etc) and fuel requirements. When used on formed tracks, very few signs of bilbies were observed. For example, 500 km of swept tracks surveyed over five nights on two occasions revealed only one set of bilby tracks. A small number of diggings, tracks and burrows were found by people walking through the bush, but the sample size was limited by the number of available observers and the distance they could cover per day. People searching on foot was not sufficiently well documented to allow a quantifiable comparison of the effectiveness of this technique with using observers on horseback, but anecdotal evidence suggests it is inefficient and not feasible in areas where bilbies are in low densities over a large area.

Southgate *et al.* (2005) reported on a trial to evaluate the efficacy of three techniques to record measures of bilby activity in the Tanami Desert. Similar to our experience, they reported that using ATVs along tracks (fixed transects) was ineffective and suggested a combination of aerial survey (helicopter) and searching random, 200 m x 300 m (6 ha) ground plots on foot. Using helicopters at Lorna Glen is cost prohibitive and searching on foot is very time consuming and physically demanding if large areas are to be searched routinely. We needed to evaluate an alternative efficient and cost-effective method to monitor signs of bilby activity (tracks,

diggings, scats, burrows) as a surrogate for relative abundance and distribution.

This note reports on the efficacy of using observers on horseback surveying belt transects cross-country to record signs of bilby activity as a basis for determining bilby occurrence and relative abundance, as measured by levels of activity. As a means of transport, horses are likely to be advantageous due to their lower environmental impact than off-road vehicles, their ability to negotiate difficult terrain including thick scrub, recently burnt and long unburnt spinifex, and they provide an elevated observation platform.

METHODS

Two stock horses were transported by float to Lorna Glen for the duration of the survey (28–31 March 2011) and stabled in the disused cattle yards near the homestead. The animals were fed a weed-seed-free diet from three days prior to arriving to reduce the possibility of introducing new weeds to the property (Van Weyenberg *et al.* 2006).

A trial survey area of ~2800 ha was delineated on a map around one of the bilby release sites, with the release site being near the centre of the sample area. Aerial photographs of the trial area enabled an assessment of the dominant soil and vegetation types covered by the transects. Vegetation mainly comprised of spinifex (*Triodia basedowii* and *T. melvillei)*) over red sandy soil with patches of mulga (*Acacia anuera*) and soft grasses over red loamy soils, making tracks, diggings and burrows readily observable. Observations were more difficult in harder, stony country. Using an access track as a baseline, a series of six cross-country belt transects varying in length from 3.35 to 6.35 km were surveyed as a sample of the survey area. The transects were 1.65 km apart and oriented north–south to minimise sun glare and maximise track visibility. For safety, two experienced observers on horseback rode within visual contact of each other, usually not more than 40 m apart. The observers followed a pre-determined bearing and carried a GPS to navigate, to measure distance travelled and speed, and to log the location of signs of bilby activity. The observers recorded signs that were confidently identifiable as bilby. Based on the observer's experience and knowledge, burrows and diggings were identified by their size and shape; other clues such as associated tracks and scats assisted with identification. Very old diggings that were highly weathered and which could have been by varanids, bilbies or rabbits, were not recorded.

Signs of bilby activity were recorded as burrows, diggings or tracks. We defined burrows and diggings as being 'old' or 'fresh' (active) according to whether they were made, or showed signs of use as evidenced by soil disturbance and tracks, before or after the last rainfall event, which was 12 days prior to the start of the survey. Tracks were separately recorded if they were not associated with diggings or burrows. During the survey, the opportunity was taken to record conspicuous signs of other animals such as feral cat, wild dog/dingo and echidna, but these are not discussed here.

From horseback, clear, unimpeded observation distances either side of the transect to the soil surface (hence, signs of bilby activity) varied depending on the vegetation height and cover. Following a pilot trial we decided on conservative visual sample distances (survey width) of 15 m for burrows and diggings and 5 m for tracks, so only signs of activity that fell within these distances either side of each observer were recorded. We estimate that, on average, the riders were approximately 15 m apart (abreast), which gave a combined average visual sample swathe, or belt transect sample size, of ~45 000 m^2 and 15 000 m^2 for burrows and tracks respectively for each 1 km of transect.

RESULTS AND DISCUSSION

From horseback, we were readily able to observe diggings, tracks and burrows of bilbies (and signs of other animals). The stock horses easily negotiated the terrain, moving across country at an average speed of 4.4 km/hour. In open country, they walked at 6.8–7.4 km/hour, but slowed to 3.3–3.7 km/hour in more difficult terrain.

The total length of transect surveyed in this trial was 27 km of which ~20 km (74%) comprised spinifex sand plains and dunefields that had been burnt ~8–9 years previously. Of the remainder ~3.5 km (13%) was older, long unburnt spinifex and ~3.5 km (13%) was scattered shrubs, small trees and soft grasses associated with palaeochannels and stony plains. While the trial was conducted over a three day period, due to other constraints and activities, actual riding time to cover the total 27 km of transect was only 6.1 h. Once operational, two observers on horseback could comfortably survey 25 km per day.

Monitoring bilby activity from horseback had a number of advantages. Horses were able to negotiate all terrain encountered including sand dunes, recently burnt and long unburnt spinifex, thick scrub, mulga groves, rocky outcrops and calcrete. Observers did not need to 'steer' the horses and so could concentrate on observing. Being on horseback provided an elevated observation platform enabling a relatively wide swathe to be visually searched. The horses were capable of walking long distances whilst maintaining an acceptable speed, and, based on a visual assessment, had a relatively low physical impact on vegetation and soil compared with ATVs and other motorised transport. Footprints left by the horses were obliterated by subsequent rain.

Based on the conservative visual sample widths described above the total area sampled or searched was 121.5 ha for burrows and diggings and 40.5 ha for tracks. This equates to 4.3% and 1.4% of the total survey area for burrows and diggings, and tracks, respectively. Bilby activity recorded along the belt transects is summarised in Table 1, but burrows were of most interest in this survey. Ten fresh or active and nine old or inactive burrows were recorded. On the basis that the belt transects were representative of the broader survey area, this equates to one fresh bilby burrow per 12.15 ha, and either a fresh or an old burrow every 6.39 ha of survey area. Therefore, the estimated number of fresh bilby burrows in the 2849 ha survey area is 234.

As the primary objective of this trial was to test the efficacy of using observers on horseback to survey for

Table 1 Summary of bilby activity observed along horseback transects.

Transect	Survey time (h)	Distance (km)	Active burrows	Old burrows	Fresh diggings	Old diggings	Fresh tracks	Detection efficiency
Day 1 T1 & T2	2.9	12.7	7	2	3	0	4	5.5
Day 2 T3 & T4	1.7	7.6	0	2	0	3	0	2.9
Day 3 T5 & T6	1.5	6.7	3	5	3	5	3	12.7
Total	6.1	27	10	9	6	8	7	6.6

For the purpose of this trial, only two transects were surveyed per day over three days. Fresh activity refers to soil disturbance after a rainfall event 12 days before the survey. Detection efficiency is number of sightings of bilby activity per hour of searching.

signs of bilby (and other animal) activity, the actual survey effort (time and distance) was relatively small. Based on 27 km of survey transect, the distribution of bilby burrows along a transect was scattered or random rather than clumped, with the exception that no burrows were recorded on the harder substrates associated with palaeochannels and stoney plains. Burrows and diggings were observed in long unburnt and recently burnt spinifex, and while diggings were easier to see in recently burnt spinifex, burrows were seen equally well in both seral stages. The frequency with which signs of bilby activity were encountered (activity sightings per hour of observation) varied considerably between the transects (Table 1). This was probably due to the varying substrates, with bilby activity confined to deeper sandy or loamy soils that are easier to dig, and to the direction of dispersal of re-introduced animals. For example transects 3 & 4 had least activity (Table 1) and were the most westerly transects, suggesting few bilbies had moved in this direction since their release. Fresh signs of activity such as diggings and burrows are evidence of bilby presence, but they do not reveal the actual number of bilbies (Lavery & Kirkpatrick 1997; Moseby & O'Donnell 2003). However, burrow density is likely to be indicative of the relative abundance of bilbies, but further research is required to establish this relationship.

We recorded relatively few bilby diggings (Table 1) possibly because we recorded diggings only if we were confident that they were bilby rather than other species. Alternatively, given the good season and abundant fresh plant growth, the bilbies may not have been digging as much for food.

Measured in terms of area searched per unit of time, observers on horseback travelling cross-country proved to be a more efficient technique for searching for signs of bilby activity than observers on foot. Southgate *et al.* (2005) reported that it took 57 days for three observers on foot to search for signs of activity in 164 randomly located 6 ha plots. While it is not clear whether this included travel time, this equates to a productivity rate of ~17 ha searched per day, or ~6 ha per day per observer. Under operational conditions (rather than trial) two observers on horseback working 30 m apart could comfortably search 120 ha per day (4.5 h actual search time) for burrows (20 km long x 60 m wide belt transect), or 60 ha per day per observer. Other comparisons with

trails reported by Southgate *et al.* (2005), such as detection efficiency, are not meaningful given differences in terrain and habitat, observer experience, and that we were surveying a recently re-introduced population rather than a 'wild' population.

CONCLUSIONS

Whatever the method of transport (foot, aerial, horseback) local habitat and terrain conditions and observer experience and skill are important factors determining detection efficiency and reliability. However, using observers on horseback to survey for signs of bilbies and other animals that are difficult to trap or spotlight proved to be an effective, low impact method of transport and a good observation platform in this landscape. While diggings and tracks can be recorded, we suggest recording burrows is more useful because they are more distinctive, relatively easy to see, are persistent in the landscape and may be a better indicator of bilby abundance. Using the number or density of fresh burrows as an index of bilby abundance, repeat surveys for burrows should enable the trend in the relative abundance of bilbies to be monitored. This trial was too small to undertake a statistically rigorous analysis of whether a sample size of 4.3% of the survey area is adequate. Further operational surveys including increasing the sampling effort to ~6% will build a dataset to enable analysis.

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Hydrated body-fluid osmolality values for species of *Cyclorana*

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A hydration procedure for frogs is described that involves two episodes of bladder emptying and rehydration prior to determining hydrated body-fluid osmolality values. The mean osmolality of hydrated *Cyclorana* species was similar for *C. australis* (lymph 225 mOsm/kg, urine 31 mOsm/kg), *C. longipes* (lymph 211 mOsm/kg, urine 40 mOsm/kg), *C. cultripes* (lymph 213 mOsm/kg, urine 14 mOsm/kg)and *C. platycephala* (lymph 214 mOsm/kg, urine 21 mOsm/kg). These standardised values provide a baseline for comparison with dehydrated and active frogs. Using this method the investigator can accurately determine standard mass (the mass of a hydrated frog with an empty bladder), which requires two or more weight measurements.

KEYWORDS: burrowing frog, *Cyclorana*, hydration, lymph, osmolality, plasma, urea, urine.

INTRODUCTION

Determining the hydrated state of a frog is important in providing a baseline or comparison value in studies relating to hydration state. This is essential in studies of the effect of dehydration on the concentration of osmolytes (Shoemaker 1964; Hillman 1978), locomotor performance (Beuchat *et al.* 1984; Preest & Pough 1989) and dehydration tolerance (Thorson 1955; Hillman 1980). Various authors have compared hydration of frogs in response to salinity (Gordon *et al.* 1961; Ruibal 1962a), periods of underground aestivation (McClanahan 1972; Katz 1989; Cartledge *et al.* 2006, 2008; Reynolds *et al.* 2011) and seasonal availability of moisture (Reynolds & Christian 2009).

The genus *Cyclorana* includes 13 species of fossorial hylids (subfamily Pelodryadinae) in northern and arid Australia (Tyler & Knight 2009). Species of *Cyclorana* undergo seasonal dormancy, have capacious bladders and develop relatively impermeable cocoons (Withers 1993, 1995; Christian & Parry 1997; Tracy *et al.* 2007; Reynolds 2011). This study details a standard hydration procedure and the resultant body-fluid osmolality values obtained for four species of *Cyclorana* from monsoonal northern Australia. The procedure also allows the investigator to measure standard mass (SM), i.e. the weight of a hydrated frog with an empty bladder (Ruibal 1962b).

MATERIALS AND METHODS

All animals were obtained from the northern monsoonal region of the Northern Territory of Australia: *Cyclorana australis* from within a 50 km radius of Darwin (12°27'50"S, 130°50'30"E), *C. longipes* from Mickett Creek (12°24'28"S, 130°56'43"E) east of Darwin, *C. cultripes* from between Daly Waters (16°15'15"S, 133°22'12"E) and Newcastle Creek (17°15'45"S, 133°27'10"E), and *C. platycephala* from 5–15 km south of Dunmarra (16°40'46"S, 133°24'46"E). Animals were returned to the laboratory and maintained in ventilated containers (18 x 12 x 8 cm) and supplied with moist paper towels. They were fed 1–

2 times a week when active. The last feed was at least five days before the procedure. Two groups of *C. australis* were used. Members of the first group (n = 12; sampled February 2008) were kept in the laboratory for 3–4 weeks and the second group for 3–4 months (n = 6; sampled July 2009) before the hydration procedure. *Cyclorana longipes* (n = 5; sampled March2009) were maintained in the laboratory for 1–2 months, *C. cultripes* (n = 4; sampled March 2008) for 4–5 days, and *C. platycephala* (n = 9; sampled April2009) for 1–2 months.

The hydration procedure required three consecutive days. At each weighing, excess fluid on the skin was removed by blotting with paper towels and the animals were weighed to the nearest 0.01 g on an electronic balance. On the first day each frog was weighed, the bladder fluid was drained, and then the frog was reweighed (SM0). The frogs were placed in a tilted container with aged tap water at one end (water access) and were left overnight. The following morning they were weighed, and the first standard mass measurement (SM1) was made by applying pressure on the area overlying the bladder, and by cannulating the cloaca with a piece of thin flexible tubing, or in smaller frogs (*C. longipes* and *C. cultripes*) with a pipette tip. After removing the bladder fluid, the frog was weighed and placed in shallow water (~5 mm depth) in a level container so that the ventral surface was in contact with the water. The next day SM was measured a second time (SM2), and lymph and urine were collected as the samples for analysis. The lowest mass recorded was used as the SM. Lymph was extracted from the femoral lymph sac with a fine gauge needle (Terumo U-100 1.0 mL 27G) as described by Reynolds *et al.* (2009). Osmolality was measured with an Advanced Instruments (freezing-point) Micro-Osmometer (Model 3300) and the BUN reagent (Thermo Electron TR12015) spectrophotometric assay was used to measure urea concentrations. One-way ANOVA was used to test the equality of means.

RESULTS

The osmolality of plasma and urine for the two groups of *C. australis* were similar (plasma osmolality $F_{1,16}$ = 0.32, P = 0.58; urine osmolality $F_{1,16}$ = 1.63, P = 0.22) so the values

were pooled for further analyses. Body fluid osmolality was similar for the four species of *Cyclorana* when hydrated (Table 1). Differences between species for the measured parameters were not significant with the exception of urine osmolality (plasma osmolality $F_{3,31}$ = 2.77, P = 0.058; plasma urea $F_{3,19}$ = 1.16, P = 0.35; urine osmolality $F_{3,32}$ = 3.37, P = 0.03; urine urea $F_{3,20}$ = 2.35, P = 0.10). Paired comparisons of means (Tukey HSD) for the significant result failed to detect a difference between species. Across all individuals of all species urea contributed 0.5–4.8% of plasma osmolality and 29–72% of urine osmolality.

Standard mass (range) was 26.5–60.1 g for *C. australis*, 5.4–9.7 g for *C. longipes*, 7.7–11.2 g for *C. cultripes* and 14.9–56 g for *C. platycephala*. Generally the mass after the bladder was drained the first time (SM0) was greater than the SM1 and SM2 measurements (Table 2). The exception was *C. cultripes* where SM0 was the lowest standard mass measurement. The SM2 measurement was noticeably lower in *C. platycephala*. The standard mass obtained after an overnight period in water (SM2) was the lowest mass for 14 of 18 *C. australis*, 4 of 5 *C. longipes*, 0 of 4 *C. cultripes* and 8 of 9 *C. platycephala*. Between measurements there was an increase in mass due to storage of fluid (water) in the bladder.

Table 1 Body fluid osmolality (mOsm/kg) and urea concentrations (mmol/L) for species of *Cyclorana* following the hydration procedure.

Species		Lymph	Urine
C. australis	Osmolality	225.2±15.2 (18)	31.3±13.7 (18)
	Urea	6.3±3.7 (6)	22.3±14.7 (6)
C. longipes	Osmolality	211±7.5 (5)	40±28.3 (5)
	Urea	4.4±1.8 (5)	25.2±21 (5)
C. cultripes	Osmolality	212.8±3.7 (4)	14±2.2 (4)
	Urea	3.7±0.8 (4)	7.6±3.3 (4)
C. platycephala	Osmolality	214.1±11.2 (8)	21.3±5.4 (9)
	Urea	5.8±2.5 (8)	12.1±4.4 (9)

Values are mean ± standard deviation (sample size in brackets).

Table 2 Variability in initial bladder empty mass (SM0), standard mass with water access (SM1) and standard mass for frogs in water overnight (SM2) on consecutive days for species of *Cyclorana*.

Species	n		SM0	SM1	SM2
C. australis	18	Mean	1.26	-0.28	-0.98
		SD	1.25	0.98	0.69
C. longipes	5	Mean	0.22	0.09	-0.31
		SD	0.14	0.10	0.21
C. cultripes	4	Mean	-0.32	0.18	0.15
		SD	0.33	0.23	0.22
C. platycephala	9	Mean	2.00	0.54	-2.54
		SD	1.85	1.14	2.11

The values are the average of the measurement (in grams) minus the average of the three values for each individual frog. SD, standard deviation.

DISCUSSION

Amphibians attain a fully hydrated state when water is readily available, generally from moist soil or directly from surface water. The mean plasma osmolality for *Cyclorana* species in this study varied from 211 to 225 mOsm/kg, and urine osmolality was from 14 to 40 mOsm/kg. Urea comprised <5% of the solute concentration of plasma and up to 72% of that of urine. In comparison, body-fluid osmolality values for hydrated frogs from the literature are generally higher, with some exceptions (Table 3). The methodology used to hydrate frogs will affect the osmolality values obtained, so that placing frogs on moist soil overnight (Cartledge *et al.* 2008) may be insufficient to reduce the osmolality of the body fluids to fully hydrated levels. Konno *et al.* (2005) also found that plasma osmolality was higher (241 mOsm) on moist soil with water access than when frogs were hydrated in water (222 mOsm). Similar hydrated osmolality values were obtained for *Rhinella marina* by Reynolds & Christian (2009). Hence, I conclude that it is necessary to place frogs in water overnight to ensure full hydration.

On the basis of this study and others where frogs were placed in water rather than on moist soil or with access to water, the plasma osmolality of anurans when hydrated is ~210–230 mOsm (Table 3). Reported values for *Bufo viridis* are consistently high and this species may be an exception. McClanahan (1972) did not provide details of his hydration procedure other than to indicate that he followed the method of Ruibal (1962b) in determining standard weights, hence I cannot comment on the value for *S. couchii*. Osmolality is useful as an index of hydration state when compared with hydrated values (Minnich 1982), and frogs that become dehydrated in the dry season attain osmolality values that are clearly elevated (plasma >300 mOsm) when compared to hydrated frogs (Reynolds & Christian 2009; Reynolds *et al.* 2011). Similarly, plasma osmolality of the aquatic frog *Xenopus laevis* is ~230 mOsm when hydrated and exceeds 400 mOsm when dehydrated (Hillman 1978).

Species of *Cyclorana* in monsoonal northern Australia are largely terrestrial in the wet season and fossorial in the dry season (Tyler *et al.* 1983; Tracy *et al.* 2007; Reynolds 2011). *Cyclorana australis* and *C. longipes* spend extended periods in the water or on saturated soils during the breeding season (S J Reynolds unpubl. data),

Table 3 Plasma osmolality (mOsm/kg) and urea concentrations (mmol/L) reported for a selection of hydrated anuran species.

Species	Osmolality	Urea	Reference
Scaphiopus couchii	301	33	McClanahan (1972)
Bufo viridis	332	22	Shpun *et al*. (1992)
Notaden nichollsi	266	20	Cartledge *et al*. (2006)
Neobatrachus aquilonius	220	7	Cartledge *et al*. (2006)
Cyclorana platycephala	250	4	Cartledge *et al*. (2008)
Cyclorana australis	222	12	Reynolds (2011)
Rhinella marina	228	15	Reynolds & Christian (2009)
Rhinella marina	222	9	Konno *et al*. (2005)

and *C. platycephala* is largely aquatic during wet weather (Robinson 1989; McMaster 2006; S J Reynolds unpubl. data). These periods in wet environments are likely to result in full hydration as outlined in this study. In addition, during the early stages of burrowing, mean plasma (214 mOsm) and urine (35 mOsm) osmolality of *C. australis* were similar to hydrated values reported here (Reynolds *et al.* 2011). Fossorial frogs are fully hydrated and have full bladders in these initial stages of burrowing.

The osmolality and solute composition of anuran lymph is generally equivalent to blood-plasma (Reynolds *et al.* 2009) and various authors (Ruibal 1962a; McClanahan 1972) have used plasma or lymph interchangeably. This is due to high endothelial permeability and comparatively rapid fluid exchange in frogs (Hillman *et al.* 2004). Thus, the hydrated values reported here are comparable with previous studies.

The procedure described here involves emptying the bladder twice prior to determining body-fluid osmolality. However, larger frogs (>100g) may require an additional bladder emptying episode to ensure that excess waste products (principally urea) are voided. Some consideration must be given to the condition of the animal prior to the procedure, as dehydrated frogs will accumulate osmolytes in the plasma (Shoemaker 1964; Hillman 1978). The condition of the body fluids prior to the hydration procedure will differ in the wet and dry season, and the osmolality of frogs obtained from burrows will vary with duration underground (Cartledge *et al.* 2008; Reynolds *et al.* 2011). Feeding prior to the hydration procedure may also influence solute concentrations, because urea will accumulate in the body fluids after a high protein meal. By allowing a period of five days following feeding and emptying the bladder twice, the frogs become fully hydrated. The two groups of *C. australis* tested had similar plasma and urine osmolalities, which suggests that the results obtained using this method are repeatable.

In addition to hydrating the frogs this procedure provides an accurate estimate of standard mass, which requires two or more measurements (Ruibal 1962b). Species of *Cyclorana* accumulated fluid in the bladder overnight, and excess solutes were removed when the bladder was emptied. In general the SM2 measurement was the lowest, suggesting that the frogs were fully hydrated at this time.

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